12670 High Bluff Drive
San Diego, California 92130
Tel: +1.858.523.5400 Fax: +1.858.523.5450
www.lw.com

LATHAM & WATKINS LLP

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VIA EMAIL & OVERNIGHT MAIL

Taly Jolish, Assistant Regional Counsel
U.S. EPA, Region 9
75 Hawthorne Street (ORC-3)
San Francisco, CA 94105

Re: United Heckathorn: Montrose's Comments on the Draft FFS

Dear Taly and Rachelle:

On behalf of Montrose Chemical Corporation of California, this letter responds to EPA's February 24, 2015 letter regarding issuance of the draft Focused Feasibility Study (the "Draft FFS") at the United Heckathorn Site (the "Site"). Montrose appreciates EPA's willingness to maintain a collaborative relationship and allow Montrose an opportunity to comment on the Draft FFS.[1] To that end, Montrose engaged Exponent Consulting and Anchor QEA to review the Draft FFS and assess its conclusions and suitability for use in remedy selection. The Exponent technical report is attached hereto as Attachment A, and the Anchor technical memorandum is attached hereto as Attachment B. Both reports conclude that the Draft FFS is critically flawed in significant ways, such that it would be imprudent to finalize the report in its current state.

Certain of the Draft FFS's key technical and legal deficiencies are described in greater detail below. Without a critical reanalysis, the Draft FFS is inadequate to select an effective and efficient remedy for the Site that meets the criteria of the National Contingency Plan ("NCP").

I. THERE ARE SIGNIFICANT TECHNICAL LIMITATIONS TO CRITICAL INPUTS RELIED UPON BY THE DRAFT FFS

The Exponent report and the Anchor memorandum include technical comments that EPA should review and respond to as part of its revision process for the Draft FFS. Detailed below

[1] Please note that, although Montrose is submitting comments on the draft FFS, it denies that it has any liability in connection with the Site.

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are two major deficiencies in EPA's post-1997 remediation investigation that serve as critical inputs to the Conceptual Site Model ("CSM"), which provides the framework for developing the amended Remedial Goals ("RGs") and for developing and evaluating remedial alternatives.

A. Source Control

Active remedial alternatives for the Site should not be analyzed without first understanding the sources and pathways of contamination and ensuring those sources are controlled. Indeed, EPA guidance states that "[i]dentifying and controlling contaminant sources typically is critical to the effectiveness of any Superfund sediment cleanup." *See* USEPA, *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*, 2005, p. 2-20. Yet the CSM asserts that "Dredging residuals are the primary source of DDT" in the Channel, without adequately evaluating the litany of potential sources EPA has already identified but failed to effectively characterize or control. These sources include without limitation: (1) pipes and conveyances from the upland area to the Channel (including those that are subtidal or terminate behind sheetpile or rip rap), (2) sediment in pockets in the riprap and contaminated embankment soils from the upland area, (3) the City of Richmond municipal outfall at the head of the Channel (including contaminated residual sediment in the uplands storm sewers), and (4) the upland cap. [2]

Moreover, EPA is impermissibly putting the remedial "cart" before the source control "horse" in directing that the City of Richmond's outfall pipe at the head of the Channel be analyzed *after* the remedial action is complete. The municipal drain, and residual sediments further upgradient in the storm sewer system, must be evaluated *prior* to the remedy to determine if it is indeed a continuing preferential pathway for dieldrin, DDT, and other contaminants of concern. Any other course of action would be inconsistent with best practices and may lead to ineffective remediation. An ongoing source identification problem is potentially fatal to effectively analyzing and weighing remedial alternatives for the Channel, and presents the potential for remedy failure due to recontamination from uncontrolled sources.

B. Dry Weather Modelling and Sampling

The Draft FFS acknowledges the limitations in certain of its inputs from sampling and modelling that only occurred during dry weather conditions, a deficiency noted in Exponent's January 23 technical memorandum to EPA. For example, in attempting to characterize whether various pipes and outfalls were potential ongoing sources, EPA did not have its consultants inspect or sample the pipes during wet weather conditions. *See FFS* 3.2.2.1 ("[T]he pipes and outfalls have not been inspected or sampled during wet weather conditions.") Without capturing

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[2] Other potential sources that have yet to be effectively characterized include the potential impact of other upgradient pesticide formulators and manufacturers, the effect of maintenance and other dredging operations in the inner Richmond Harbor, and post-remedial storm events (including the 13-year storm that occurred on December 31, 2005) which may have led to episodic inflows of sediment from the storm drain systems and other piping and laterals. *See* Attachment A Section 2.10. Each of these potential sources require further analysis.

the episodic flow that accompanies wet weather conditions, the sampling is incomplete and insufficient to properly inform remedy selection.

In addition, the simulation period for the Sediment Transport Study, which was specifically incorporated into the Draft FFS and served as a basis for the CSM, was limited to a 34-day dry-season period. Important sediment processes occur during wet-weather conditions, yet EPA admittedly made no attempt to quantify or estimate sediment loadings that might occur during these episodic flow events. This failure to simulate the wet periods that are *most important* to the spatial and temporal distribution of sediment and contaminants means that the models are not a reliable basis for analyzing remedies that must be effective during both dry and wet conditions.

II. THE SUGGESTED REMEDIAL GOAL IS BASED ON UNREALISTIC ASSUMPTIONS AND MUST BE RECONSIDERED

The revised RGs for protection of human health and ecological receptors are based on a number of unrealistic and overly conservative exposure and toxicity assumptions from draft risk reassessments that were performed by CH2MHILL in 2010.[3] The ecological RG reassement for fish is seriously flawed for a number of reasons, including: (1) fish tissue samples in the channel were not paired with representative sediment concentrations, (2) as a result, the bioaccumulation models are unreliable and imprecise, and (3) the Fish-based DDT toxicity reference value was inappropriate because none of the studies involved fish species in the Channel and the selected values were not developed for sediment assessment or management. Similarly, for birds, the data used to model bird diet are inappropriate and area use was not considered, implying that the receptor population obtains its entire diet from the Lauritzen Channel when in fact these birds typically forage over a much larger area. Moreover, there is no basis to assume that birds would prefer to forage in the Channel -- a narrow, noisy, lighted and very active industrial waterway. Finally, the human health Risk Based Concentration ("RBCs") were based on unrealistic assumptions regarding the fish consumption rate from a study among the Laotian community in West Contra Costa County, the majority of which only fish in freshwater areas, with an assumption that 50% of the fish consumed within this community comes from the Site. Indeed, the site is inaccessible for fishing, and even if it were accessible, there is no evidence that this heavy industrial waterfront would be an attractive daily fishing spot for an angler for 30 consecutive years, as assumed by EPA.[4] As a result, each of the RGs calculated from the 2010

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3 Importantly, it does not appear that EPA or CH2MHILL addressed comments from Shell and Geoystenc that identified significant issues that needed to be addressed prior to completing the documents, including the failure to consider the central tendency exposure, inappropriate data usage and assumptions, and the use of an ill-conceived "shot gun" method at modeling bioaccumulation to ecological receptors. A copy of Shell's comments are attached hereto as Attachment C. These comments should be addressed prior to utilizing conclusions from the 2010 risk reassesements as the basis for developing new RGs.

4 Even if all of EPA's assumptions were true, including that (i) there is risk in eating fish from the Channel (which there is not), (ii) the site is accessible to anglers (which it is not), (iii) an angler would otherwise fish there every day for 30 years (which they would not), and so on, such alleged fishing could easily be addressed through institutional and engineering controls (such as "no fishing" signs).

risk reassessments are unnecessarily and unjustifiably conservative, leading to recommendations of unnecessary cleanup.

When realistic and scientifically justifiable assumptions are substituted for the worst-case assumptions used in the 2010 risk reassessments, none of the sediment RBCs for DDT exceed the original RG from the 1994 ROD (590 µg/kg). *See* Attachment A, at Table 4 (noting corrected sediment RBC for Shiner Surfperch should be multiple times higher than 400 µg/kg). It appears likely that piscivorous birds, not fish like the shiner surfperch, are the ultimate theoretical risk driver for DDT at the Site.[5] As a result, a defensible Site-specific area use factor should be developed in connection with setting a revised cleanup level to protect birds from DDT exposure. For illustrative purposes, a RBC of 1000 µg/kg can be used to estimate an appropriate level when accounting for actual area use (the 2010 RBCs assume 100 percent area use, which is unrealistic given that, for example, the average daily forage radius for Forster's terns has been reported at 4.9 km from nest sites).

While the limited data and time available to review the Draft FFS was insufficient for Montrose's consultants to conduct a fully revised risk reassessment, it is vital that a significant critical reconsideration of the RBC calculations be part of the final FFS. Without alteration, the 2010 RBCs are unsupportable as RGs.

III. NON-DREDGING ALTERNATIVES WARRANT DETAILED EVALUATION

The Draft FFS summarily rejects all available technologies beyond dredging with little or no analysis. Scant rationale is provided for scoring of rejected alternatives, and, in many cases, the scores appear inconsistent with successful implementation of remedial technologies at similar sites and the conclusions of reports incorporated in the Draft FFS. The Draft FFS should provide a more thorough exploration of the potential advantages and disadvantages of in situ treatment – including the placement of an activated carbon layer throughout the channel, engineered capping, confined disposal of sediments within the channel, and various combinations of all three. These technologies can be equally as effective as dredging, without the added environmental and community impacts or increased costs associated with a dredging-centric remedy.

Primarily, the Draft FFS does not adequately justify why in situ treatment technologies, including activated carbon amendment, were not carried through for actual consideration in the vast majority of the channel. Activated carbon was given low scores for effectiveness (FFS Table 5-3), even though carbon amendment is incorporated into proposed remedial alternatives to a limited degree, and is described elsewhere in the report as effective and promising, with a 90 to 99 percent reduction in apparent bioavailability of DDT in Site sediment (see FFS Section 2.8). These site-specific results are consistent with successes at other sediment sites with in-situ treatment using activated carbon, including at other active industrial waterways. *See* Patmont et. al. (attached hereto as <u>Attachment D</u>); Ghosh et. al. 2011 (attached hereto as <u>Attachment E</u>).

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5 Similarly, using a 90th percentile fish consumption rate from APEN (1998) and a modified fish fraction from the site of 10%, the resulting human health tissue RBC is 8.59 mg/kg (wet wt) in edible tissue, a value 10-fold higher than the overly-conservative value calculated by the flawed assessment of CH2M Hill.

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Indeed, EPA's own guidance regarding the use of carbon amendments for in situ remediation notes that "[u]nlike other remedies, amendments applied to the surface sediments have some potential to adsorb contamination from continuing sources as well as from sediment sources," a particularly relevant consideration at this Site considering the ongoing source issues. EPA 2013, at p. 11. Various procedures and products have been developed to facilitate the placement such that activated carbon can be administered to the sediment, including proprietary products that are specifically designed to sink in the water column, while also providing additional resistance to being resuspended by erosive forces, scour, and other disturbances. Once bound to the carbon, the resulting reduction in bioavailability of the organic contaminants is not dependent on maintenance of an intact layer, making sediment scour and redistribution much less of a concern.

In tandem with more appropriate RGs, as discussed above, activated carbon and the other highlighted technologies can be effective at reducing the spatially weighted average concentration ("SWAC") to levels that would meet the selected RBCs. Even using the inappropriate RG developed in the Draft FFS (400 µg/kg), these technologies can be effective in reducing the Channel SWAC to cleanup levels. Further consideration of these alternatives can also lead to the development of efficient hybrid approaches that include some combination of carbon amendment, engineered capping, targeted hotspot dredging, and/or onsite confined disposal. In addition, each of the remedial technologies Anchor proposed for further analysis satisfy EPA's evaluation criteria for analyzing alternatives. *See, e.g.*, *Guidance*, at 6-3 (the nine evaluation criteria include overall protection of human health and environment; compliance with ARARs; long-term effectiveness and permanence; short-term effectiveness; reduction of toxicity, mobility, or volume; implementability; cost; state acceptance; and community acceptance).

IV. EPA FAILED TO ADEQUATELY WEIGH AND SCREEN REMEDIAL ALTERNATIVES IN THE DRAFT FFS

A. EPA Effectively Evaluated Only One Remedy

EPA only analyzed slight variations of the same remedial alternative—dredging—in the Draft FFS. Given the complexity of the Site and the technical effectiveness of other alternatives, EPA should more carefully analyze non-dredging alternatives. Indeed, EPA's own guidance dictates that non-dredging alternatives be carried through for further analysis.

The goal of an effective feasibility study is to analyze a sufficient range of alternatives depending on the scope and characteristics of the site. *See* 40 C.F.R. § 300.430(e)(2). For source control actions, like here, the range of alternatives should include, as appropriate: (1) an alternative that removes hazardous substances or contaminants to the maximum extent feasible; (2) alternatives that, at a minimum, treat the principle threats posed by the site through varying degrees of treatment; (3) one or more alternative that involve little or no treatment; and (4) a no action alternative. *Id*. at (e)(3). While "the typical target number of alternatives carried through screening usually should not exceed 10," the alternatives carried through should still adequately preserve the range of remedies initially developed. *See Guidance*, at 4-26. Critically, variations of the same remedial procedure do not amount to independent alternatives as required by the NCP. *See Sherwin-Williams Co. v. City of Hamtramck*, 840 F. Supp. 470, 478 (E.D. Mich. 1993)

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(holding that City's work plan improperly considered and analyzed only varying degrees of soil excavation).

Although EPA developed a range of initial alternatives in its technology screening evaluation, those carried through for further analysis in the Draft FFS do not preserve the initial range. Rather, the Draft FFS considers four alternatives: the statutorily required no action alternative (immediately disregarded) and three dredging alternatives. The only variance between the three alternatives is the amount of dredging that occurs in the Northern Head of the Channel – an area that makes up only 8,000 cubic yards of the 66,000 cubic yards EPA seeks to remediate. Thus, the **sole** remedy considered for the majority of the Lauritzen Channel (the West Side and the East Side making up roughly 88% of the remedial footprint) is dredging. Only analyzing dredging alternatives cannot give EPA a meaningful opportunity to assess the efficacy of any of the other alternatives it initially developed. *See, e.g.*, *Versatile Metals, Inc. v. Union Corp.*, 693 F. Supp. 1563, 1582 (E.D. Pa. 1988) (determination of the efficacy of remedial actions should not be made "in a vacuum").

Accordingly, EPA has failed "to gather information sufficient to support an informed risk management decision regarding which remedy appears to be most appropriate for [the Site]," (*Guidance*, at 1-3) and further analysis of alternatives that were prematurely screened out is required. The Draft FFS is invalid for failure to analyze a sufficient range of alternatives.

B. EPA Significantly Underestimates the Costs of Dredging, Leading to an Erroneous Presumption In Favor of Dredging-based Remedies, and Avoidance of Internal Remedy Review

Although absolute accuracy of cost estimates is not essential, EPA guidance gives a desired range of accuracy for evaluating costs of potential remedies. *See Guidance*, at 4-24. At the alternative screening stage, EPA expects an accuracy range of -50 to +100 percent, which means "for an estimate of $100,000, the actual cost of an alternative is expected to be between $50,000 and $200,000." *See* EPA, *A Guide to Developing and Documenting Cost Estimates During the Feasibility Study*, at 2-5 (July 2000). At the detailed analysis stage, the expected accuracy should range from -30 to +50 percent. *Id*. at 2-6.

At its detailed analysis stage, EPA presented a ROM cost estimate of $22,711,303 for Alternative 4—dredging of the entire Channel. [6] Applying the range of -30 to +50 percent, the actual cost of Alternative 4 should be between $15,897,912 and $34,066,955. However, based on a critical reanalysis of the cost drivers in EPA's estimate, the actual costs associated with Alternative 4 could easily exceed $35 million, with the same level of accuracy.[7] *See* Attachment

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[6] A similar comparison of costs would apply to Alternatives 2 and 3 because the majority of costs are attributable to dredging 88% of the channel – a figure that remains constant through all three alternatives.

[7] For example, an ongoing environmental dredging project in San Diego Bay, roughly the same size anticipated by EPA at the United Heckathorn Site, is estimated to cost more than $40 million. Unlike the current site, the San Diego Bay sediment is being managed at a local non-hazardous landfill, rather than an out-of-state

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B at p. 18. EPA's estimate may even fail to fall within the accuracy range expected at the screening stage, which would accommodate costs as high as $45 million.

Underestimating the costs associated with dredging apparently led EPA to favor remedies with heavy dredging footprints and, in any event, resulted in an inaccurate representation of the feasibility of the dredging alternatives that were considered. Because the actual cost of dredging will likely far surpass EPA's modest estimates, it is incumbent on EPA to consider more cost-effective alternatives that are scientifically appropriate for the Site.

Moreover, EPA's policy on remedy review states that any remedy estimated to cost over $25 million is subject to review by EPA's Remedy Review Board or regional remedy review team. *See* Memorandum re National Remedy Review Board Criteria Revision and Operational Changes, OSWER Directive 9285.6-21 (Sept. 4, 2014). By significantly underestimating costs, the FFS would appear to avoid further internal critical review by EPA teams established for that purpose. That level of scrutiny is even more important here, in light of the past extensive dredging conducted at the site, potential ongoing sources, and limited evaluation of technologies other than extensive dredging.

V. CONCLUSION

Montrose is deeply concerned with the deficiencies highlighted by Exponent and Anchor, and the potential for the Draft FFS to lead to potentially unnecessary cleanup. Montrose is hopeful that EPA will seriously consider the comments submitted by Montrose and other stakeholders at the Site, and incorporate those comments into a revised FFS. Through collaborative effort—such as the upcoming June 2015 technical meeting—Montrose believes the selection of a scientifically appropriate, legally defensible, and cost-effective remedy for the Site is attainable. In turn, Montrose reserves the right to supplement these comments as additional matters arise that would be useful for EPA's consideration and helpful towards finalizing the FFS and issue a Proposed Plan.[8]

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hazardous waste landfill. Hence, it is reasonable to assume that dredging at the United Heckathorn site would be equal to (or more like, much greater than) the costs for the San Diego Bay project.

8 For example, Montrose and its consultants have yet to review any documents in response to Montrose's January 26, 2016 request for supplemental data. Montrose only received access to a portion of the requested documents on May 20 (2 days before close of the comment window period). Further documents being sent on a "thumb drive" have not yet been received. Therefore, Montrose has not had an opportunity to review these documents or incorporate the results of the review into the comments submitted herewith. As initially noted, these documents remain critical to conducting a thorough assessment of the conclusions reached in the various technical reports relied upon by EPA, and to address possible limitations in those studies, which were explicitly incorporated into the draft FFS.

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Please do not hesitate to contact me if you have any questions or would like to discuss.

Very truly yours,

Kelly E. Richardson
of LATHAM & WATKINS LLP

cc: Rachelle Thompson, EPA

Attachment A

Technical Memorandum

Comments on United Heckathorn Superfund Site Draft FFS—February 2015

Prepared for

Joe Kelly, President
Montrose Chemical Corporation of California
600 Ericksen Avenue NE, Suite 380
Bainbridge Island, WA 98110

Prepared by

Exponent
15375 SE 20th Place, Suite 250
Bellevue, WA 98007

May 22, 2015

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Comments on United Heckathorn Superfund Site Draft FFS—February 2015

We have reviewed the subject document, as well as many cited supporting reports and studies, and have the following technical comments.

Executive Summary

The draft FFS is critically flawed in a number of significant ways, any of which is sufficient to call into question the conclusions and render recommendations regarding the scope and type of required additional remedy invalid. Major deficiencies of the draft FFS include the following:

- The revised remedial goals (RGs) for protection of human health and ecological receptors are based on a number of inappropriate exposure and toxicity assumptions. The resulting RGs are therefore unnecessarily and unjustifiably conservative, leading to recommendations of unnecessary cleanup. Furthermore, the justification and basis of these goals is poorly documented in the draft FFS, and the rationale behind decisions about risk tolerance and exposure assumptions is entirely missing. In setting revised RGs, EPA has relied entirely on sediment RBCs developed by a 2010 risk reassessment that is both technically flawed and inappropriately biased for purposes of risk management. Many assumptions in the exposure models from which the RBCs are derived are screening-level in nature and are not realistic.
- The remedial alternatives evaluated are far too limited and narrow. The three alternatives included in the draft FFS, all of which are based on extensive dredging throughout the Lauritzen Channel, are scarcely different from each other. EPA failed to adequately evaluate several obvious alternatives such as monitored natural recovery (MNR), enhanced natural recovery (ENR), or more extensive use of activated carbon to sequester sediment pesticides *in situ*. We demonstrate, using simple area-weighted average exposure reduction calculations, that beneficial uses can be protected with far smaller dredging footprints than those proposed in the draft FFS, if realistic exposure assumptions and *in situ* technologies are used.
- Identification and quantification of potential ongoing sources of pesticide contamination to the Lauritzen Channel is incomplete. As a result, conclusions about the relative significance of ongoing sources are poorly justified, and the actual potential for recontamination following remedy implementation cannot be adequately assessed.
- In particular, the extensive stormwater system which drains into the Lauritzen Channel has not been adequately assessed for contamination, integrity, or even

fully described. It is not possible to accurately assess the potential significance of stormwater outfalls as a historic or ongoing source of sediment contamination.

- The sediment transport model and pesticide mass balance calculations used to develop the conceptual site model (CSM) and evaluate effectiveness of remedial alternatives are flawed, inadequately validated, and poorly documented. Of particular concern is the reliance of the transport model on a single month of dry-season data. Many aspects of the transport model, including initial and boundary conditions, calibration, and validation are inadequately explained in the publically available reports, and cannot be fully evaluated[1].

For each of the above reasons, which are further described in detail below, we believe the draft FFS, in its present form, is inadequate to inform the selection of an effective and efficient remedy for the Site. Our recommendation is that EPA reassess RGs to be consistent with the best available science and realistic exposure assumptions (see discussion of possible RG revisions in our comments on FFS Chapter 6). Even given the issues with EPA's methods described below, we further recommend that EPA more completely and rigorously assess critical inputs to the remedy selection process which are deficient and expand the range of remedial alternatives considered and evaluated before finalizing the FFS or making any remedial decisions. In some cases (e.g., sediment transport modeling), refinement of existing analyses would require additional data. In other cases, existing data have not been properly or fully evaluated. Where appropriate, we have suggested examples of the type of reanalysis that is possible with existing data to support a reasonable and protective remedy.

Specific Comments

We have the following comments on specific elements of the draft FFS, organized sequentially and referenced to the FFS chapter and section.

Chapter 2 Post-Remediation Investigations

This chapter discusses all of the site-specific investigations and data upon which EPA relies to form the CSM for the site, which is then used to identify and evaluate the selected remedial alternatives. Summary reports for most of these investigations have been previously published. A few key investigations (i.e., Source Identification Study, Tier 1 and 2 Sediment Transport Studies, DDT Fate and Transport Studies) are appended to the draft FFS. The following are

[1] In our January 23, 2015 preliminary technical memorandum, we developed a list of data and information from EPA required to support a more thorough review of the conclusions reached in the various EPA study reports, including additional pollutant concentration data and detailed information describing the hydrodynamic and sediment modeling studies. We understand this request was then restated to EPA by Montrose on March 31. EPA's consultant CH2MHill ultimately delivered a portion of the documents we requested to Montrose on May 20 (2 days before the close of the comment period) and we have not had sufficient time to review those documents. Accordingly, our review was circumscribed by the available data, and we were not able to conduct as thorough an assessment of the conclusions reached in the various technical reports relied upon by EPA and explicitly incorporated into the draft FFS.

comments on specific studies and lines of evidence with regard to their suitability or limitations to support the conclusions of the draft FFS.

Section 2.1 Post-Remediation Biomonitoring

Mussel sampling for pesticide bioaccumulation monitoring purposes was conducted 10 times between 1998 and 2013. While the draft FFS briefly describes this line of evidence and cites it as evidence that food web exposure has been demonstrated, little interpretation of the data record is offered. In fact, there are notable trends in the bioaccumulation data and previous analysis that should be reviewed and fully evaluated in the draft FFS.

The first 5-year review report noted an initial, transient post-remediation increase in pesticide bioaccumulation levels, with decreasing mussel tissue concentrations in 1999–2001, even though the remedial objectives for dieldrin and DDT concentrations in water and sediment had not been met at that time (USEPA 2001). The second 5-year review report, which included biological monitoring data through 2003, documented a general continuing decline in DDT levels in mussel and fish tissue, with sediment and water remedial objectives being met in some but not most other areas of the Channel (USEPA 2006). The third 5-year review report added biological data from 2007 and 2009, which show an increase in mussel tissue DDT residues, back to pre-remedial levels (USEPA 2011). Taken as a whole, the bioaccumulation data record suggests a change in Site conditions between 2003 and 2007, leading to a reversal of the observed decrease in biological uptake of DDT attributed by EPA to success of the remediation at the time of the second 5-year review. The reasons for this are unclear but should be thoroughly assessed prior to attempting any additional remedial action. An evaluation of events during this time period (i.e., weather events, changes in Channel use, construction, maintenance dredging, stormwater data) could offer important clues about the reasons for the bioaccumulation increase as well as the performance of the original remedy and importance of sources of recontamination. In particular, a review should be undertaken of major rainfall events over the post-remedial time period (2000 to present), and an examination of how apparent sediment concentrations may have been influenced by episodic stormwater discharges, based on sediment data trends over this same period. For example, a 50-year storm event occurred in Contra Costa County on December 31, 2005. Effects of the surge of accumulated sediment from storm drains could be reflected in the 2007 sediment data and contemporary bioaccumulation data, especially near stormwater outfalls. A year by year review of such major precipitation events could help assess the significance of stormwater as a source during the post-dredging period of interest.

Section 2.8 Carbon Amendment Treatability Study

A site-specific bench-scale study of *in situ* sediment treatment using activated carbon was performed in 2007 (Tomaszewski et al. 2007). EPA acknowledges the promising results of the study and high likelihood of effectiveness in the reduction of DDT bioavailability under site conditions, noting that “The ground, reactivated carbon resulted in a 91 percent reduction in SPMD uptake after 1 month and a 99 percent reduction in SPMD uptake was achieved after 26 months (using 3.2 percent application rate). The effectiveness of reactivated carbon for sequestering DDT was not diminished over 26 months of treatment, demonstrating that DDT was not rereleased from the activated carbon." (FFS, p. 2-6). However, EPA fails to include *in situ* treatment as one of the primary remedial alternatives in the FFS. No explanation is offered for the relegation of *in situ* treatment or carbon amendment to use only in the activated cap proposed for the northern head of the Channel, and as a source control measure. Given the promising site-specific results of the bench-scale study, more extensive use of carbon amendment, either as a standalone remedial alternative or in conjunction with hotspot removal should have been evaluated.

In-situ remediation of chlorinated bioaccumulative compounds such as DDT and PCBs has been shown to be an effective remedy in numerous pilot studies and in full-scale applications. In addition, using activated carbon treatment technologies can limit the community impact of remediation while reducing the risk of exposure. USEPA (2013) discussed the applicability of activated carbon amendments, and USEPA headquarters is currently encouraging the use of activated carbon for *in situ* remedies that include a variety of application methods. This reflects the strong scientific consensus concerning the value of such methods (Ghosh et al. 2011, Patmont et al. 2014). For example, the Department of Defense has been demonstrating the efficacy of *in situ* remediation with activated carbon for DDT compounds in sediments at Aberdeen Proving Ground. Demonstrations have also been carried out by the Navy in harbors and bays, and a substantial activated carbon field demonstration project is being planned for San Francisco Bay this summer. The State of Delaware recently implemented a successful full-scale *in situ* activated carbon application (essentially bank to bank) in a tidal system known as Mirror Lake, which has resulted in significant improvement (http://www.dnrec.delaware.gov/News/Pages/New-DNREC-video-Mirror-Lake-One-year-later-finds-significant-improvement-in-lakes-health.aspx).

The experience to date indicates that *in situ* remediation can be implemented in open water areas without additional capping, so long as the technical details of such an approach account for the physical characteristics of the area as well as desired goals. Unlike a cap, which is a physical barrier designed to keep contaminated sediments in place, the use of activated carbon relies on vertical mixing of the carbon material and contaminated sediments to reduce bioavailability and exposure. Once bound to the carbon, the resulting reduction in bioavailability of the organic contaminants is not dependent on maintenance of an intact layer. Sediment scour and redistribution is thus much less of a concern with a properly designed and implemented activated carbon remedy than a cap, sand cover, or any other form of physical sequestration.

Despite the documented success of activated carbon treatment, EPA fails to include any *in situ* treatment option as one of the primary remedial alternatives in the draft FFS. We urge USEPA to reconsider this position, and give this alternative due consideration as part of the FFS.

Section 2.10 Source identification Study

The recent Source Identification Study (SIS, CH2M Hill 2014, FFS Appendix B) is the report cited by the draft FFS as the authoritative statement on potential sources of pesticides in Lauritzen Channel sediments, water, and biota. It builds upon earlier Phase I, II, and III source identifications and is an important input to the CSM. Seven potential sources were evaluated by the SIS. However, the SIS evaluation of at least four major potential ongoing sources of sediment contaminants is inadequate to support the CSM and remedy selection and/or is inadequately considered in formulation of the CSM by EPA. Each of these sources is discussed briefly below. It is critical that all four of these sources should be more adequately assessed and quantified prior to final remedy selection or implementation.

Storm Drains and Other Outfalls

The draft FFS concludes that, with the possible exception of municipal and Levin Richmond Terminal Company (LRTC) property stormwater outfalls, pipes and outfalls are not a significant ongoing source of contaminants to the Lauritzen Channel. This dismissal of possible ongoing outfall sources is not supported by the SIS or available data. Pre-emptive cleaning of major stormwater laterals is proposed as part of each of the three evaluated remedial alternatives (see FFS, Section 3.2.5). Several historical lines of evidence regarding the potential of stormwater outfalls as sources of contaminated sediment are ignored or inadequately considered by the draft FFS.

A narrative of a 2001 site inspection included in first 5-year review report states that the "Lauritzen Channel has numerous outfall pipes, including interceptor outfalls and City of Richmond outfalls" (USEPA 2001, p. 15). Conditions within these storm drain systems have not been well characterized, and the potential impact of other upland pesticide formulators and manufacturers (e.g., Calspray) have yet to be addressed. Thus, it is unknown whether stormwater or other discharges have been or may continue to be a significant source of sediment and contaminants to the Lauritzen Channel.

It appears that sediment was not sampled in storm drains following the original remediation until 2007 (CH2M Hill 2011, Attachment 1, Table 1). Because conditions have been dry in recent years, pipes and outfalls "have not been inspected or sampled during wet weather conditions" (CH2M Hill 2011, p. 3-3). According to annual reports, "…occasional minor sedimentation [is] observed within the storm drains" (CH2M Hill 2011, p. 6-2), indicating transport of soils. In 2008, sampled sediments within storm drains had detected concentrations "up to 52 mg/kg" of DDT (CH2M Hill 2011, p. 6-6), which were attributed to historical operations and lack of cleaning. Reports indicate that "to date [March 2014], the municipal storm drains have not been cleaned out; therefore, the stormwater sampling will not be conducted and the cleaning of the storm drain system will be included in the evaluation of remedial alternatives in the FFS" (CH2M Hill 2014, p. 2-2). As noted in the SIS, "Stormwater discharges from the municipal storm drain at the head of the Lauritzen Channel were to be sampled as part of this source identification study after the residual sediments in the storm drain system had been removed. However, these sediments have not yet been removed, so the potential for the municipal storm drain system to act as an ongoing DDT transport pathway in the future cannot be evaluated. If the residual sediments are removed prior to completion of the FFS, then stormwater sampling may be performed, to verify whether or not discharges from the municipal storm drain system are an ongoing source of contamination to the Lauritzen Channel. Otherwise, development of the remedial alternatives should address this potential ongoing source" (CH2M Hill 2014, p. 6-1). The recommendation of the SIS has not been fully followed by EPA in the draft FFS.

While the draft FFS does recommend pre-emptive cleaning of storm drains, it is important to understand the historical and current roles of the storm drain system as a potential ongoing source, especially because previous characterization has been inadequate and incomplete. For example, reports indicate that "Storm drain sediment sampling was also performed by EPA's START contractor in 2012 to support a potential emergency removal action. Due to cost implications, the removal action was placed on hold, and the sampling report was not finalized; therefore, the data are not included in this evaluation" (CH2M Hill 2014, p. 6-2). The reports also indicate that "…the structural integrity, invert elevations, and hydraulic connections could not be determined for all drains because of the large amount of residual sediment in the system" (CH2M Hill 2014 p. 6-1), and the reports describe cracks in piping and water infiltration (CH2M Hill 2014, p. 6-3). Further, none of the reports have addressed the potential effect of post-remedial storm events, which may have led to episodic inflows of sediment from the storm drain systems and other piping and laterals. It would be prudent to more completely evaluate available information related to storm drain and outfall discharges before proceeding with remedy selection to avoid selection of an inappropriate or premature additional remedy before the recontamination potential is fully understood.

Recent reports acknowledge that concentration "bounce-back" occurred in several interceptors over the years, indicating the importance of characterizing source mechanisms prior to remediation (CH2M Hill 2014, p. 6-3). Understanding whether this failure was the result of additional pollutant sources from storm water or other outfall discharges is important to assessing the need for and timing of additional remedial measures.

Upland Areas

Beyond embankment soil erosion, the possibility of significant ongoing sources from upland areas is not evaluated by the SIS or acknowledged in the draft FFS. EPA appears to be relying on the finding of the third 5-year review report, which perfunctorily concluded that "[t]he remedy implemented at the upland areas of the United Heckathorn Superfund Site is protective of human health and the environment, due to capping of contaminated soils which has eliminated human health exposure pathways and prevented erosion. Routine inspection and monitoring assures the protectiveness of the upland remedy at the Site…" (CH2M Hill 2011, p. viii). These conclusions appear to be based on annual reports that document the implementation of the operations and maintenance plan and found that "the upland cap is determined to be uncompromised and functioning as intended" (CH2M Hill 2011, p. 6-1). However, it does not appear that runoff over the capped upland areas was sampled, or that pollutant concentrations have been measured in "occasional minor sedimentation observed within the storm drains" (CH2M Hill 2011, p. 6-2). Moreover, based on photographs included in the third 5-year review report that show visible cracks in the upland cap, the integrity of the cap seems at best unclear (see USEPA 2011, p. 20–21). Without additional documentation and data, it appears to be premature to conclude that the upland area is not contributing sediment and pollutant loads to the marine areas, or that drains associated with upland areas contribute to recontamination of sediments. The possibility of contribution from upland area runoff to post-remedial DDT sediment concentrations was also raised by Anderson et al. (2000), who noted that post-remedial sediment DDT to DDD concentration ratios were intermediate between ratios measured in pre-remedial sediments and upland soils. The authors concluded that "This suggests that post-remediation contamination may have come either from an upland source or from one where the timing or conditions under which metabolic alteration of DDT differed from those of the pre-remediation sediments” (Anderson et al. 2000, p. 885). A thorough evaluation of potential inputs of contaminants from upland runoff or erosion should be added to the draft FFS.

Subtidal or Obscured Outfalls and Seeps

The draft FFS CSM dismisses the potential of subtidal or obscured outfalls and seeps as significant ongoing sources of sediment contaminants. However such hidden pathways are known to exist. As noted by the SIS, “… other pipes and conveyances that are not visible may exist (i.e., features that terminate behind rip rap or sheetpile, or are subtidal). Any of the identified or unidentified pipes and conveyances could have and may still act as preferential pathways for the transport of DDT from the upland area to the Lauritzen Channel, particularly adjacent to the former plant site and former train scale area where highly contaminated soils and groundwater still exist” (CH2M Hill 2014, p. 9-1). Clearly, it would be prudent to characterize these potential sources and understand their importance as an ongoing source of sediment and contaminants to the receiving waters before proceeding with additional remedy selection and implementation.

Embankment Soils

The potential for embankment soil erosion to be an ongoing source of sediment contaminants is acknowledged both by the SIS and the draft FFS CSM. However, no further assessment of this pathway or incorporation into remedy evaluation or assessment is included. Site surveys have noted areas of erosion (“erosion hotspots”) and seeps in the past. In addition, the existence of “preferential pathways” for contaminant migration has been suspected (CH2M Hill 2014, p. 3-2), but it is unclear whether such pathways have been characterized. As with other potential sources, the magnitude of these sources has not been well characterized, and it is not clear whether these sources have been addressed. For example, “Evidence of soil erosion was observed during the site surveys performed in 2012. Erosion under the sheet pile wall, observed as approximately 1- to 2-foot voids, was noted at the north end of the eastern bank of the channel. These features were noted between bent -37 and the head of the channel. Sink holes and exposed cap material were also observed on the Levin property in the vicinity of bent -24 and T-8.5” (CH2M Hill 2014, p. 3-3). It appears that an embankment soil erosion hotspot near bent +3 to bent -3 was not addressed during work in 1990–1993, or during 2002–2004 (CH2M Hill 2014, p. 3-2). Although a seep at T-8.5, “an ongoing source of DDT contamination to the channel” (CH2M Hill 2014, p. 3-2), was sealed in 2003, it is not clear whether the seal was effective, or if it is routinely inspected and maintained, nor can it be determined whether other similar seeps exist or have been sealed, or whether significant amounts of pesticides were released before it was sealed. Finally, “… historical embankment soil and sediment data indicate that erosion of contaminated embankment soils on the northern and eastern sides of the channel is an ongoing source of contamination to the Lauritzen Channel. However, the magnitude of the source is difficult to quantify because most of the embankment is lined with sheetpile, rip rap, and/or concrete, with only localized areas of exposed soil subject to erosion.” (CH2M Hill 2014, p. 3-5). Several embankment soil samples were opportunistically collected during the 2013 sediment characterization study and were found to contain elevated levels of both dieldrin (up to 380 µg/kg) and total DDT (up to 14,100 µg/kg) (see FFS Table 3-2). The significance of this source should be fully characterized and evaluated before proceeding with selection of a remedy that only addresses sediments.

Section 2.11 Sediment Transport Study

The Tier 2 Sediment Transport Study (STS, Sea Engineering 2014, FFS Appendix D) is incomplete or inadequate in many respects, with several significant shortcomings detailed below. Accordingly, EPA's modeling does not currently provide the information needed to support evaluation or selection of a remedy. Because sediment dynamics at the site play such an important role in understanding the reasons for failure to maintain the original remedial objectives and in predicting the performance of future remediation, we recommend that EPA significantly enhance the existing transport model.

The model did not account for wet-weather conditions.

The primary flaw in the STS reports is that the simulation period in the hydrodynamic and sediment transport models was limited to a 34-day dry-season period from June 4 to July 9, 2013. However, sediment resuspension is typically greatest during storm events when wind and wave conditions transfer the greatest amount of energy to the sediment bed. Sediment loads from land surfaces to receiving waters are also greatest during storm events. The numerical model simulations, therefore, are incomplete, because the simulations do not capture the important processes that occur during wet-weather conditions and do not attempt to quantify or estimate sediment loadings to the model domain that occur during episodic flow events. In addition, the monitoring period is clearly not justified, given the conclusion that "[t]he total daily averaged sediment flux over the 34-day mooring deployment period was near zero kg/s at both locations. The near zero sediment flux was observed during a one-month dry period. Overall net accumulation in San Francisco Bay typically occurs during the wet fall and winter periods" (Sea Engineering 2014, p.17). The failure to simulate the wet periods that are most important to the spatial and temporal distribution of sediment and contaminants means that the models are not a reliable basis for selecting a remedy (or remedies) that must perform during both dry and wet conditions.

The model excluded physical processes that are important for accurately estimating contaminant concentrations.

The STS reports do not describe the hydrodynamic and sediment transport processes used in the respective models and, hence, are incomplete from the perspective of understanding the model documentation and the model review process. The reports describe the use of the Environmental Fluid Dynamics Code (EFDC), which includes various constitutive equations and formulations that can be selected by the user. Processes that influence cohesive sediment transport include advection, dispersion, aggregation (flocculation), settling, consolidation, and resuspension. The reports do not identify the processes and formulations that were implemented in the model. For example, the two sediment size classes simulated (10 µm and 51 µm) fall into the cohesive class range given the relatively high fraction of mud in most of the surface samples. Consequently, the settling velocities of these sediments are susceptible to aggregation (flocculation) in the water column; flocculation will result in settling velocities that are likely to differ from those calculated using the Cheng (1997) formulation. Furthermore, it is not clear how the model resuspends the two sediment size classes from the sediment bed or how these sediment size classes are tracked in the sediment bed and in the water column. Finally, it appears that anthropogenic activities, such as scour from vessel movement, dredging activities, and outfall discharges, were not included in the model, possibly leading to a failure to identify all important transport mechanisms responsible for elevated sediment DDT concentrations.

The initial and boundary conditions to the model were inadequate.

Accurately modeling hydrodynamics and sediment transport requires appropriate initial conditions (which are used to describe the starting point of the model runs) and boundary conditions (which are used to characterize conditions at model boundaries). Both are inadequately described and may have been inadequately specified. For dynamic simulations, initial conditions need to be set up for all dependent variables. For the hydrodynamic model, these variables include salinity, temperature, and velocity in all seven sigma layers for all grid cells in the model domain. For the sediment transport model, initial conditions include the fractions of the two sediment classes simulated, the dry density, and sediment erodibility (erosion rate function and critical shear stress) with depth within the sediment bed for all grid cells in the model domain. Suspended sediment concentrations also need to be specified for all seven sigma layers in all grid cells in the model domain. As described in Sea Engineering (2014, 24–28), a Sedflume analysis was conducted for 10 cores in the Lauritzen Channel. Results showed that the erosion rates were highly variable. Because of the limited number of samples and their variable erodibility, it appears that the Sedflume tests could not be used to set up the initial bed sediment conditions in the Lauritzen Channel. The relevance of the Sedflume tests, therefore, is limited to assessment of site-specific erosion, and the tests do not provide the required spatial discretization (horizontally and vertically) for use in the sediment transport model. If data are limited for setting up the initial conditions, then the effectiveness of the model, in its current state, is likewise limited for supporting the goals of the draft FFS.

It also appears that the boundary conditions to the hydrodynamic and sediment transport models neglected important components. The hydrodynamic model was forced by only two boundary conditions—namely (i) the water levels at the Richmond Inner Harbor Tidal Station, which were applied to the southern boundary of the model domain and (ii) wind data from the National Oceanic and Atmospheric Administration (NOAA) Station at Richmond, which were applied uniformly over the entire model domain. Hydrodynamic boundary conditions that were not incorporated in the model include (i) meteorological boundary conditions other than wind speed and direction, (ii) freshwater flows from all outfalls, (iii) non-point surface runoff, and (iv) groundwater flows. Similarly, boundary conditions to the sediment transport model that were not incorporated into the model include sediment loading at all outfall locations, from non-point sources, and at the tidal boundary; the sediment loading would also need to specify the concentration of each of the two sediment size classes simulated. Absent specification of relevant boundary conditions, the hydrodynamic and sediment transport processes cannot be simulated realistically. The models, in their current state, appear therefore to be unreliable for supporting sediment management decision-making.

Inadequate Model Calibration and Validation

Typically, a model is calibrated by adjusting model parameters so that modeled and measured data match for a given time period. Models are then validated by simulating an additional time period using the model parameters from the calibration and comparing the model output to measured data. Finally, sensitivity/uncertainty analyses are typically provided to evaluate the sensitivity of the model to changes in key model parameters. Here, however, calibration was limited, and validation was not performed. It appears also that sensitivity/uncertainty analyses of model input parameters were not performed.

The hydrodynamic and sediment transport models were not adequately calibrated or validated to past or monitored conditions and, hence, cannot be expected to serve as a viable tool for predicting future conditions. For the hydrodynamic model, it appears that only water level (stage) was used to compare model predictions to measured data; validations of model output for other hydrodynamic parameters were not presented. Specifically, it appears that model-data comparisons of water levels were carried out at only one of two Acoustic Doppler Current Profiler (ADCP) locations (i.e., at the mouth of the Lauritzen Channel), which is minimal at best. Model-data comparisons of velocity were not depicted graphically; instead, the report states that “the low signal-to-noise velocities in the system did not facilitate direct model comparison” (Sea Engineering 2014, 57) and alludes to modeled tidal velocities being consistent with analytical solutions of tidal velocities based on the tidal prism. Model-data comparisons of salinities and temperature were also not performed. The study reports did not describe what hydrodynamic model parameters were used to calibrate the model but instead states, “The water levels described above were used as the primary calibration and validation metrics” (Sea Engineering 2014, 57). Although calibration parameters for hydrodynamic models typically include the bottom roughness and the mixing coefficient, the report does not substantiate the conclusion that “[o]verall, the model was insensitive to adjustments in background eddy viscosities and bottom roughness, typical of similar systems, giving confidence in the model for the applications below” (Sea Engineering 2014, p. 57).

Similarly, it appears that the sediment transport model was not calibrated or validated. Model-data comparisons of the spatial and temporal distribution of suspended solids would have provided insight on model performance. Calibration parameters that are relevant to the sediment transport model include sediment size class modeled, distribution of grain sizes in the model domain, settling velocities, and erodibility characteristics such as erosion rates and critical shear stress for erosion, and dry density. The inadequacy of the calibration and validation effort severely limits the reliability and usefulness of the model, and calls into question the validity of any future remedy selection based in any significant way on the findings of this model.

A rigorous sensitivity analysis of the model parameters was not performed.

A sensitivity/uncertainty analyses requires that each calibration parameter (e.g., initial conditions, upstream and downstream boundary conditions, bottom roughness, mixing coefficients, and sediment size and erodibility characteristics) be perturbed above and below their optimized values to evaluate model response to hydrodynamic circulation patterns and sediment concentrations in the water column and sediment bed. Absent a thorough sensitivity/uncertainty analysis, the model cannot be used reliably as a predictive tool.

In conclusion, it appears that model performance was not evaluated with sufficient rigor so as to develop confidence in the model. Hindcasting and mass balance analyses could have been conducted to provide additional confidence in the modeling tools. In addition to the inadequacy of the model calibration and validation efforts and the lack of sensitivity/uncertainty analysis, there was no attempt to perform hindcast simulations to assess the reliability of the model using known or estimated inputs from the past, to see how well the model reproduces known conditions. A hindcast simulation for the period from completion of remediation to current conditions could have provided confidence in the model, and (as discussed in greater detail below) might have provided important insight into the performance of the prior remedy. Finally, the modeling study did not perform a diagnostic analysis for sediment mass balance for the simulation period, to show that sediment mass is conserved in accordance with the equation, Input – Output = Storage. The lack of a hindcast simulation and mass balance diagnostic analysis undermines the credibility of the models.

Chapter 3 Conceptual Site Model

The CSM is the working model of sediment contaminant sources, inventory, and exposure upon which all analyses and conclusions of the draft FFS are based. In updating the CSM, EPA attempted to summarize, integrate, and synthesize all the lines of evidence, including those commented upon above. However, the draft FFS includes very little synthesis or interpretation of previous studies. Most sections of this chapter simply restate or directly quote other documents. In several cases, shortcomings of prior studies are overlooked and conclusions are simply accepted without further explanation or justification. In other cases, the CSM inadequately or incorrectly considers key information or misrepresents the significance of prior studies, rendering the analyses and conclusions based on the CSM invalid.

Section 3.2 Sources of Contamination

In summary fashion, the draft FFS briefly describes each of the seven potential pathways identified in the SIS as potential pesticide sources. The conclusion reached is that “Dredging residuals are the primary source of the DDT mass currently found in the Lauritzen Channel.” (FFS, p. 3-1), and all other sources are arbitrarily dismissed (FFS, p. 3-2: “Additionally, none of the other potential sources that were identified appear to be contributing sufficient masses of DDT to the Lauritzen Channel to account for the concentrations currently seen in the channel sediments”). Such sweeping conclusions about source identification and control are not justified and are even contradicted by the summary of other potential sources in this section of the draft FFS:

- “However, other pipes and conveyances that are not visible may exist (i.e., features that terminate behind rip rap or sheetpile, or are subtidal) and *may still act as preferential pathways for the transport of DDT* from the upland area to the Lauritzen Channel, particularly adjacent to the former plant site and former train scale area where highly contaminated soils and groundwater still exist.

Additionally, *the pipes and outfalls have not been inspected or sampled during wet weather conditions*." (p. 3-2, emphasis added)

- "Although the shoreline is largely armored with riprap and sheetpile, fine-grained sediments are present in pockets in the riprap and soils are eroding from under the sheetpile in some areas; therefore, erosion of contaminated embankment soils on the northern and eastern sides of the channel is an ongoing source of contamination to the Lauritzen Channel." (p. 3-3)
- "The City of Richmond municipal outfall at the head of the Lauritzen Channel cannot be fully evaluated as an ongoing source of contamination to the Lauritzen Channel until the DDT-contaminated residual sediments within the storm drain system are removed." (p. 3-3)
- "The stormwater monitoring data collected for the storm drain system that serves the upland cap on the LRTC property indicates that the system is functioning as designed, with only infrequent *direct discharges* to the Lauritzen Channel." (p. 3-4, emphasis added)

EPA guidance states that, "[i]dentifying and controlling contaminant sources typically is critical to the effectiveness of any Superfund sediment cleanup." (USEPA 2005, p. 2-20). The CSM's flawed discussion of sources is inappropriate to serve as the "Framework for developing the amended RAOs and RGS…and for evaluating and developing remedial alternatives," (FFS, p. 3-1), and further analysis by EPA to implement and evaluate the effectiveness of source control actions should be completed *before* finalizing the CSM and, ultimately, the evaluation of alternatives for the Channel.

Section 3.4 Sediment Transport Processes

The STS model output is the primary basis for this component of the CSM, and all findings of the STS are accepted without further interpretation as an accurate model of sediment transport in the Lauritzen Channel. The STS conceptual site model is incomplete and unreliable in explaining sediment and associated contaminant transport.

The model output was used to develop the sediment transport CSM specific to the STS. However, the CSM was based primarily on flawed modeling results and limited field studies conducted during the approximately 1-month dry-season period. As a result, the CSM is incomplete and unreliable in explaining sediment and associated contaminant transport and distribution in the Lauritzen Channel. Given the modelling flaws and other limitations associated with the STS reports (see discussion above), the CSM is incomplete and unreliable in explaining sediment and associated contaminant transport and distribution in the Lauritzen Channel. For example, ADCP measurements and modeling results from the dry-season period were used to show that the Lauritzen Channel is a low-energy environment and a sediment sink in the absence of ship traffic. Again, limited ADCP data were used to show that maximum tidally induced bed shear stresses were only slightly above the critical shear stresses measured in the Sedflume analysis, to support the assertion that "tidal currents do not play a significant role in mobilizing sediment in the Lauritzen Channel" (Sea Engineering 2014, p.67).

As part of the CSM, the Tier II STS report presents a conceptual sediment budget. Key sediment loading sources to the system include the tidally driven inflows from San Francisco Bay and upland sources. As reported, the 34-day averaged sediment flux calculated from the ADCP data showed a net tidally driven transfer of zero. A net tidally driven transfer of zero appears to contradict the assertion in the report that, “The bay provides a constant delivery of silt and clay to the margins, including harbors” (Sea Engineering 2014, p.70). Consequently, tidally driven sediment loading was not quantified in the report, which instead states, “Had the ADCPs been deployed during winter months, increased flux from the bay may have been more apparent” (Sea Engineering 2014, p.70). Because sediment delivery from upland sources was not quantified due to the lack of data, the report estimates sediment delivery using the U.S. Department of Agriculture method, which gives a gross estimate of sediment loading based on average rainfall and watershed area. Given that high flow events resulting from high-intensity rainfall produce the most sediment loading to a system, it is unrealistic to rely on gross methods to compute sediment delivery from a watershed.

The CSM and sediment budget are, at best, conceptual in nature and do not provide insight into which sediment transport processes are most important or how these sediment processes influence the potential spatial and temporal distribution of sediment and contaminants within the study area. The available data, which indicate significant increases in both sediment and DDT mass (Sea Engineering 2014, Table 5), are difficult to reconcile with this CSM. In particular, the statements about ongoing sediment losses from the Lauritzen Channel and contaminant transport to Santa Fe Channel and San Francisco Bay are unreliable and poorly-justified. Because the sediment budgets are conceptual and do not characterize conditions during the all-important wet season and for episodic events, they are inadequate for remedy selection or in predictions of remedy performance.

Non-Pesticide Contaminants

A glaring omission in the CSM discussion, and indeed the draft FFS as a whole, is the lack of assessment or even discussion of any sediment contamination at the Site other than dieldrin or DDT. Given the period of time since the original ROD and changes that have taken place in risk assessment methodology and practice (as evidenced by EPA’s reassessment of pesticide risks at this site), it would be appropriate to assess potential beneficial use impairment for all elevated sediment contaminants and costs/benefits of any remedial alternatives evaluated in reducing impairment.

Given the long history of industrial development and activity at the Site, it is not surprising that elevated concentrations of constituents other than dieldrin and DDT have been measured in sediment, soil, and water samples. For example, concentrations of metals (e.g., arsenic, cadmium, copper, lead, zinc) and other organic contaminants (e.g., polycyclic aromatic hydrocarbons [PAHs], polychlorinated biphenyls [PCBs]) are commonly measured in environmental samples, as part of remedial actions, discharge permitting, property transactions, and routine monitoring. Elevated post-remedial concentrations of PAHs, PCBs, and chlordane in Lauritzen Channel sediments have been documented, and post-remedial concentrations were as high or higher than pre-remedial concentrations (Anderson et al. 2000). The source of these contaminants remains uncharacterized, but Anderson et al. noted that industrial activities in the channel area include shipping operations and a "variety of land-based businesses, including manufacturing, recycling, and construction," all of which are potential sources (Ibid.). In addition, sediment characterization reports and analysis performed by LRTC in connection with maintenance dredging in the Santa Fe Channel show elevated concentrations of PAHs and PCBs (among other constituents) in channel sediments (Pacific EcoRisk 2009). Finally, analysis performed by the Contra Costa Clean Water Program illustrates that PCB concentrations are elevated at certain municipal stormwater sampling locations along a conveyance that appears to ultimately discharge at the outfall at the northern head of the Lauritzen Channel (EOA 2007). Although these non-pesticide constituents are not currently the target of planned remedial activities, several of the maximum concentrations reported in sediments exceed generic chemistry benchmarks commonly used for human health and ecological risk screening purposes (e.g., NOAA ER-Ls). While not necessarily indicative of unacceptable risk or the need for action, screening benchmark exceedances may indicate the need for further evaluation. In addition, concentrations of all anthropogenic constituents, together with concentrations of dieldrin and DDT can be used in many circumstances to establish source fingerprints. For example, concentrations of metals may be higher in stormwater than in embankment sediments, and the presence (or absence) of those metals in receiving-water sediments can be used to characterize the source of those sediments and the contaminants found on those sediments. Without a site-specific risk assessment, it is unclear whether these elevated metal and organic sediment contaminants currently represent a potential impairment of beneficial uses in the Lauritzen Channel, independent of pesticide contamination.

Information on other constituents present can also contribute significantly to the understanding of pollutant fate and transport at a site. For example, concentrations of metals in sediment cores collected from the Palos Verdes Shelf were critical to understanding that DDT was biodegrading at that site—peak concentrations of metals in cores from that site remained relatively steady in cores collected over long periods of time, while concentrations of DDT in the same cores decreased over time, indicating that sediment mixing was not responsible for declining concentrations of DDT (see, e.g., Paulsen et al. 1999). If available, concentrations of additional constituents should be obtained and reviewed in order to supplement the source identification work completed to date and to put together as complete a picture as possible of the various sources of contaminants to the receiving waters at the site.

The draft FFS should include a full cost-benefit analysis of remedial options. Toward this end, EPA should analyze existing data to determine which elevated constituents are impairing beneficial uses and how any evaluated remedial alternative would mitigate existing impairment. Although other constituents are not covered by the 1994 ROD at the Site, based on our experience with TMDLs throughout the state, it is possible that additional constituents may need to be addressed. In addition to the likelihood that storm water discharges and surface runoff from industrial facilities in the area have contributed to sediment contamination (see discussions above), the Lauritzen Channel and the surrounding waterways have a long history of commercial shipping terminal use. Sediment contamination scenarios commonly associated with shipping operations include petroleum hydrocarbons from fueling and treated wood piers as well as copper and organotin loadings from the attrition of antifouling hull paints. Studies conducted in active harbors have concluded that leachate from copper-based hull coatings can be the primary dissolved copper loading source (Bloom 1995, US Navy 1998). Incorporating additional constituents into a planned remedy now would maximize the likelihood that beneficial uses will be protected by future remedial actions and protect against the failure of future remedies due to elevated levels of non-pesticide sediment contaminants.

Chapter 4 Remedial Action Objectives and Remediation Goals

The justification for and derivation of revised RGs is among the most fundamental findings of the draft FFS. Yet the derivation process is among the most poorly documented and weakest in the report. No narrative of the appropriateness, basis, inherent assumptions, or technical strengths and weaknesses of the human and ecological risk assessments used as the basis of the amended RGs is presented, and it is impossible to evaluate the proposed RGs using the information included in the draft FFS[2]. In several respects, we find the revised RGs to be based on inappropriate and unrealistic exposure assumptions, poor scientific interpretation of toxicity data, and over-simplified characterization of exposure conditions at the Site. No technical shortcoming noted in this review has a greater significance to the draft FFS conclusions or validity. We strongly recommend that the amended RGs be revised, fully explained, and justified as reasonably protective and obtainable goals. We have offered some examples below of the analyses which are missing or improperly documented or have been performed incorrectly. A full reassessment is beyond the scope of this memorandum, but EPA should perform a full reassessment using available data prior to finalizing the FFS.

Section 4.2 Summary of 2010 Reassessment of Ecological and Human Health RGs

Unlike other inputs to the remedial alternative selection process (i.e., extent of contamination, fate and transport, bioavailability), no supporting information concerning the risk evaluations that drive RG derivation is appended to the draft FFS. Two 2010 memoranda are cited as the basis of the amended RGs, one dealing with ecological risk and one with human health risk (CH2M Hill 2010a and 2010b, respectively). We have reviewed these documents and found the analyses contained in them to be severely flawed in several critical respects, invalidating the risk-based target concentrations for use as cleanup levels.

Section 4.2.2 Ecological RG Reassessment

The ecological risk reassessment (CH2M Hill 2010a) derives and tabulates a large number of potential sediment risk-based concentrations (RBCs) for consideration in risk management. Tissue RBCs for pesticides are first calculated based on either a critical tissue residue approach (for fish, shrimp, and mussels) or a food web model (prey tissue levels protective of piscivorous wildlife). Sediment RBCs are then estimated using several different bioaccumulation models (see below). The draft FFS proposes an amended sediment RG of 400 µg/kg for protection of all ecological receptors, which is the mean sediment RBCs developed for protection of shiner

[2] The two 2010 reassessment memoranda (CH2M Hill 2010a and 2010b) were not appended to the draft FFS, nor were they available on the EPA website for the UH Superfund Site. This does not meet technical or transparency standards for establishment of risk-based RGs. Only through review of documents obtained on the Envirostar database were we able to evaluate the technical validity of the amended RGs.

surfperch (CH2M Hill 2010a, Table 21). Surfperch is predicted by the reassessment to be the most susceptible ecological receptor assessed, and, therefore, surfperch sediment RBCs are predicted to be protective of all other modeled ecological receptors as well as human health. However, the derivation of the surfperch RBCs and many other RBCs in the reassessment memorandum are seriously flawed in several respects and are therefore unsuitable for direct use to develop risk-based cleanup levels, at least without further interpretation and modification.

Fish Bioaccumulation Models

The proposed amended RGs in the draft FFS for both ecological and human health are driven by fish bioaccumulation. Proposed fish tissue levels of dieldrin and DDT are stated to be protective of either fish, piscivorous wildlife, or anglers, based on RBCs from the ecological and human health risk reassessment memoranda. Target sediment concentrations of dieldrin and DDT stated to be protective of ecological or human receptors are then back-calculated from these protective fish tissue concentrations using a biota-to-sediment accumulation factor (BSAF) predicted by the site-specific bioaccumulation model for shiner surfperch, which in turn is developed in the ecological risk reassessment memorandum. The surfperch bioaccumulation models are therefore a critical underpinning of both the ecological and human health amended RGs. Unfortunately, the derivation of these models is critically flawed and significantly over-predicts measured uptake of pesticides by surfperch in the Lauritzen Channel, ultimately resulting in much lower RGs than necessary for protection of ecological or human receptors.

Shiner surfperch is one of ten fish and invertebrate species for which bioaccumulation models were developed in the ecological risk reassessment. Species-specific models were developed for mussels, bay shrimp, anchovy, jacksmelt, flatfish (includes halibut, sanddab, and starry flounder), goby, staghorn sculpin, and shiner surfperch. In addition, bioaccumulation models were developed to predict the average uptake of all benthic fish (flatfish, goby, and sculpin), all water-column fish (anchovies, jacksmelt, and surfperch), and all sampled biota. The surfperch model was selected by EPA for use in developing amended RGs, because it predicts the highest bioaccumulation of any of the models developed, and is therefore the most protective. However, it is clearly not the most representative. This worst-case biouptake assumption is itself inappropriate for the development of cleanup levels. More importantly, no critical review of the underlying data limitations or predictive ability of the surfperch uptake model was performed.

For each of the receptor species or groups listed above, three independent bioaccumulation models were developed: logistic regressions of bulk concentrations (tissue wet wt vs. sediment dry wt), logistic regressions of lipid-TOC normalized concentrations (lipid-normalized tissue vs. TOC-normalized sediment), and the output of Trophic Trace, a commercial model based on equilibrium partitioning theory. The logistic regression approach used by CH2M Hill (2010a) is technically sound. Logistic regressions were computed for paired fish and sediment concentrations across a range of pesticide levels in the Richmond Inner Harbor area. However, there are several fundamental ways in which the both the underlying data and the execution of the models were flawed.

1) Fish tissue samples in the Lauritzen Channel are not paired with representative sediment concentrations.

Biota samples used to develop the bioaccumulation models were collected from five stations in the Lauritzen, Santa Fe, and Richmond Inner Harbor Channels as well as Parr Canal in May and June of 2008. Sediments were sampled from the same areas in August 2007. While not synoptic, these data were reasonably well matched temporally. Unfortunately, in some cases, inappropriate representative sediment concentrations were matched with specific tissues samples. Association of a given fish sample with a single sediment location or sample is always uncertain or even impossible, because fish move and feed over areas of various sizes, depending on species and local habitat. Benthic fish species can be expected to more closely associated with a finite area of sediment than pelagic species, if their capture location is known. In this particular study, most fish were caught using bottom trawls. Therefore individual fish cannot be associated with a precise catch location, only with a trawl line.

Recognizing this limitation of the data, the ecological risk reassessment authors used mean sediment concentrations from the sampled areas. However, mean concentrations are not representative of exposure conditions when sample locations are unequally distributed. In environmental investigations, areas of known contamination are typically sampled at a higher density than relatively clean areas, leading to high bias in mean or median detected concentrations. To avoid such bias, the appropriate approach to represent an area with high sediment concentration variability is to use a spatially-weighted average concentration (SWAC) rather than a mean to represent typical exposure conditions across the entire area. There are a number of geospatial interpolation (i.e., contouring) techniques that can be used to develop a SWAC, but the simplest and most objective approach is Thiessen polygons, whereby each point in an area of interest is assumed to be represented by conditions at the nearest sampled location, without interpolation or averaging. The result is a mosaic of polygons of variable shapes and sizes, each surrounding one sampled location, based on the spatial distribution of the samples. The sediment SWAC for a given constituent can easily be calculated using Thiessen polygons by summing the products of each polygon area and measured concentration and then dividing that sum by the total area. Figure 1 is a Thiessen polygon map for the Lauritzen Channel, constructed using all surface sediment sampling locations from 2007. Total DDT SWACs calculated using this polygon map are shown in Table 1. The total DDT SWAC for all of the Lauritzen Channel is 7,026 µg/kg. The average value used to develop the bioaccumulation models by CH2M Hill was 10,648 µg/kg (CH2M Hill 2008, Table 1), a value more than 50 percent higher than the SWAC.

Even more importantly with respect to the ultimate use of their RBCs, neither the bioaccumulation models nor the fish sampling program in 2008 incorporate the fact that radically different sediment concentrations and exposure regimes exist in the northern and southern reaches of the Lauritzen Channel, even though this unequal distribution of sediment pesticides is one of the primary characteristics of the sediment data and should have factored prominently into the study design. As Figure 1 and Table 1 show, if the channel is bisected along Thiessen polygon boundaries into roughly equal halves, the northern reach has a DDT SWAC over five times higher than the southern reach. Some water column species, such as anchovies and topsmelt, may move throughout the entire channel and, to some degree, average their exposure over most or all of the channel. For many of the sampled fish species, including gobies, sculpin, and surfperch, which have very small home ranges, it would be inappropriate to assume that fish tissue samples from the southern reach reflect exposure conditions in the northern reach or vice-versa. By averaging sediment chemistry across the entire channel, the bioaccumulation modelers ignored the sharp gradient in exposure conditions from one end of the channel to the other. This obfuscates one of the most important and potentially informative dimensions of the site with respect to understanding exposure and bioaccumulation.

Further, designations for fish collections in the report are misleading. The 2008 fish sampling data report (CH2M Hill 2008) includes samples attributed to both designated biomonitoring stations in the Lauritzen Channel, the designations for station 303.2 (labeled “South Lauritzen”) and station 303.3 (labeled “North Lauritzen,” see Figure 1). A careful review of the sampling narrative (CH2M Hill 2008, p. 6) and the plot of GPS trawl lines (Ibid., Figure 3) make it clear that the biota samples labeled 303.2 and “South Lauritzen” were actually caught in the Richmond Inner Harbor Channel, south of Parr Canal. The trawl line was nowhere near station 303.2. This unfortunate and unexplained sampling design results in the loss of exposure gradient information that could have been obtained had both biomonitoring stations in the Lauritzen Channel actually been sampled. The sampling area associated with station 303.3 is described as follows: “Individual trawls were run for approximately 5 - 10 minutes, and extended the length of the channel, centered at historic biomonitoring Station 303.3.” (CH2M Hill 2008, p. 5). The plotted GPS trawl lines (Ibid., Figure 3) show that at least some trawls included areas of both the northern and southern reaches of the Lauritzen Channel, as described above, although the trawl line portions in the northern reach appear to be longer. As a result, it is not possible from the information provided to identify where individual fish were caught. This flawed implementation of the study results in a loss of useful bioaccumulation information. Average tissue concentrations of DDT for all biota, shiner surfperch, and benthic fishes (the fish most closely associated with sediments) are included in Table 1 on both a wet weight and lipid-normalized basis.

2) Bioaccumulation models are unreliable and imprecise.

The problematic outcome of CH2M Hill’s inadequate fish sampling design is that fish tissue samples from the Lauritzen Channel cannot be matched to any sediment concentration. As a result, they should not be considered to represent an average exposure level over the entire channel. Some of the species collected, for example staghorn sculpin, have home ranges as small as a few square meters. Further, DDT concentrations at individual sediment stations across the Lauritzen Channel vary by more than three orders of magnitude (23 to 53,765 µg/kg). The inability to match fish tissue with even a rough sediment concentration range makes the data highly unsuitable for use in a logistic regression or equilibrium partitioning model of bioaccumulation.

The bioaccumulation models developed by CH2M Hill (2010a) should be considered to have poor accuracy or predictive ability. Furthermore, data from the Lauritzen Channel exerts a high amount of leverage on the logistic regressions, because the average sediment DDT concentration paired with all Lauritzen Channel biota samples (10,648 µg/kg) is higher than any other station in the bioaccumulation study by more than an order of magnitude. This is especially true for the species which accumulate higher levels of pesticides, notably shiner surfperch (see CH2M Hill 2010a, Figure 19[3]).

The uncertainty in the bioaccumulation regressions for shiner surfperch, benthic fish, and all fish combined and the predictive ability of the logistic regression bioaccumulation models is assessed in Table 2. The range of possible DDT BSAFs that could be computed for the Lauritzen Channel is shown using various measured sediment concentrations. Measured BSAFs are the real mean values. The degree to which the models differ from measured values indicates model predictability at these concentrations. The minimum and maximum detected sediment concentrations are not realistic but are included only to bound the actual uncertainty range of BSAFs. Because we cannot determine where in the channel any biota sample was collected, the actual ratio of tissue to sediment could be anywhere in this range. A value closer to some central tendency in the sediment concentration gradient is more likely. As discussed above, the best central tendency for exposure modeling, absent any information about receptor location, is the SWAC, not a mean detected value. Due to the trawl area bias toward the northern half of the channel, the northern reach SWAC is thus the best available option.

For all biota, which is obviously the largest, most spatially averaged data set, the four BSAF estimates are in close agreement with the exception of the lipid/TOC-normalized regression model. Lipid and TOC normalization should, in theory, improve the performance of any bioaccumulation model for hydrophobic contaminants. However, this theoretical advantage depends on accurate measurement and incorporation of lipid and TOC data. All of the lipid/TOC-normalized models developed in the risk reassessment are flawed in that they all assume a sediment TOC value of 1.25 percent. In fact, in the Lauritzen Channel, the average measured TOC value is nearly twice as high (2.2%). This error results in significant divergence of the bulk concentration and normalized logistic regression models, especially at the low or high ends of the concentration spectrum.

However, for shiner surfperch, the difference between measured and modeled BSAF values is far more pronounced. This reflects the fact that the logistic regressions have poor prediction ability in the tails of the sediment concentration distribution, and this species has the highest range of measured tissue concentrations. The surfperch wet weight/dry weight model, which was used to calculate the amended RGs proposed by the draft FFS, over-predicts measured uptake by 50% at the northern reach SWAC concentration (see Table 2). The surfperch regression model is particularly unreliable and should not be used to support the draft FFS. The regression for benthic fish shows similar poor performance in terms of agreement between measured and predicted uptake on a wet weight/dry weight basis.

The predictions of Trophic Trace, which is a Gobas-type equilibrium partitioning model, is especially sensitive to data representativeness issues. Trophic Trace assumes that measured or assumed concentrations are related to each other as a function of known thermodynamic relationships (solubility and diffusion primarily). It constructs a multi-dimensional regression model that assumes all compartments in the environment are at equilibrium. This is never

[3] Note that all of the scatterplot figures of fish vs. sediment chemical concentrations in CH2M Hill (2010a) have erroneous x-axis scales, which are shifted left by an order of magnitude (x-axis values are all 10-fold too low). However, the underlying data appear to be correct.

actually true in dynamic systems like bays and estuaries. Gobas models can be calibrated to perform well in a specific environment, but this calibration requires accurate information on the co-variance of tissue and sediment concentrations. Given the uncertainties associated with spatial variability of concentrations in these data, a Gobas model is a poor choice and should not be used.

As a result of these flaws in data collection and data interpretation, the fish tissue bioaccumulation models developed in the ecological risk reassessment must be considered on the whole to be unreliable and are therefore inappropriate to use for calculating sediment RBCs without re-evaluation and modification. This has profound implications for all of the RBCs developed in the ecological risk reassessment, including those for piscivorous wildlife. All of these RBCs should be reassessed using the full range of possible BSAF values before making any remedial decisions.

3) The fish tissue-based DDT toxicity reference value is inappropriate.

The whole-body tissue-based DDT threshold estimate used by CH2M Hill (2010a) to predict adverse effects in all fish species is 0.60 mg/kg (wet wt). This value is taken from a review paper of DDT and mercury effects on fish (Beckvar et al. 2005), which tabulates both no-effect residues (NERs) and low-effect residues (LERs) from a diverse group of studies. The authors of the ecological risk reassessment took this value directly from Beckvar et al. (2005) without modification or further interpretation. This value was never intended to be a cleanup level. The objective of Beckvar et al. (2005) was to compare various methods for assessing variability in the toxicology literature for ultimate use in development of a protective tissue residue threshold to support water quality criteria development. It had nothing to do with sediment assessment or management. They reviewed toxicity studies from the published literature that reported both NER and LER values. The selected DDT value of 0.6 mg/kg is derived from a review of nine studies on adult fish, all laboratory exposures to technical grade DDT or DDE, administered via aqueous and/or dietary exposure. None of the studies involved sediment exposure or exposure to environmentally weathered DDT, and none of them were conducted on a fish species that occurs in the Lauritzen Channel area or on a species that is closely related to any fish receptor evaluated by the ecological risk reassessment. The species tested in the nine source studies of the Beckvar et al. (2005) review include three freshwater salmonids (lake trout, cutthroat trout, and brook trout), two common freshwater laboratory models (goldfish and fathead minnow), two anadromous marine salmonids (chinook and coho salmon), and one marine shallow-water species from the subtropical Atlantic (pinfish). Salmonids as a group are known to be highly sensitive to most toxicants. While typically protective, they are a poor choice as a representative marine species for risk assessment. Most of the endpoints measured are ecologically relevant (i.e., growth, lethality, or reproduction), with the exception of the goldfish study, which reported only a behavioral endpoint and should not be used at all for risk assessment or management purposes.

Most importantly, the method used by Beckvar et al. (2005) to combine the disparate endpoints and tissue concentrations from the papers they reviewed into a single protective value is a method used to derive screening levels, not cleanup levels. The DDT tissue threshold of 0.6 mg/kg is a tissue threshold effect level (t-TEL), an analog to the sediment concentration threshold effect level (TEL). A TEL is a conservative, screening level approach designed to be protective of the most sensitive members of a population or community. It is not a level at which ecologically significant adverse effects on a population are expected, nor is it a level which should trigger cleanup. The TEL concept is well established in sediment assessment. TELs were first derived from freshwater sediment toxicity studies to characterize the low end of the range of sediment chemical concentrations that affect different components of an exposed benthic invertebrate community, and are part of a two-tiered screening level that also includes the probable effects threshold (PEL). The originators describe the TEL as “Represents the concentration below which adverse effects are expected to occur only rarely” and the PEL as “Represents the concentration above which adverse effects are expected to occur frequently” (Smith et al. 1996). The method has been used by the Canadian Council of Ministers of the Environment and the State of Florida to develop ecological risk-based screening levels for sediment concentrations (CCME 1995, MacDonald et al. 1996). The t-TEL is an extension of the method from sediments to tissue concentrations. As derived by Beckvar et al. (2005), the t-TEL is the geometric mean of the median concentration in the no-effects data set and the 15th percentile concentration in the effect data set. In other words, this value is primarily a function of NERs— tissue concentrations at which no adverse effects occur. Screening level risk assessments typically make use of such values for identification of sites and exposure scenarios which require additional assessment or more site-specific data. A TEL should not be used directly as a risk-management or cleanup target. In sediment assessment, even the higher PEL value has been shown to correctly predict toxicity little more than half the time (Becker and Ginn 2008).

It should also be noted that the food web model-based risk calculations used in the ecological risk reassessment to calculate RBCs for piscivorous wildlife are all based on more appropriate lowest-adverse effect levels (LOAELs). The reason for the difference between the assessments for fish and fish-eating birds and mammals is not clear.

4) The data used to model bird diet were inappropriate.

Based on the sediment RBCs calculated by CH2M Hill (2010a, Table 21), the secondary ecological risk drivers after fish are piscivorous birds (e.g., Forster’s tern and double-crested cormorant). The lowest wildlife RBCs calculated in the 2010 reassessment were consistently for Forster’s tern, but risk to all bird species (including tern, cormorant, and surf scoter) were modeled using a dietary toxicity reference value (TRV) of 0.28 mg/kg body wt/day, a LOAEL value from Carlisle et al. (1986). The endpoint in this study was egg-shell thinning, an endocrine effect of DDT unique to birds. This is a relevant and appropriate endpoint but obviously only for females. Male and female cormorants and scoters were assessed independently by CH2M Hill (2010a) due to their different mean body weights, but this TRV has no relevance to male birds.

The tern was the avian driver primarily because of its small size (smaller animals eat more relative to their body size) and its diet, which was modeled by CH2M Hill (2010a), that contains a relatively high fraction of shiner surfperch. In fact, terns are opportunistic feeders that will take whatever small fish are available. They also eat significant numbers of small insects, crustaceans, and amphibians (see CA DFG species profile). The hypothetical diet used in the

CH2M Hill model, composed of 44% shiner surfperch, 39% water column fish (jacksmelt and anchovy), and 17% goby was based on information reported by Baltz et al. (1979), who studied prey selection of terns nesting near Elkhorn Slough, Monterey Bay, by looking at stomach contents. This high number of shiner surfperch in the Elkhorn Slough study was stated by the authors to be driven by local abundance (the highest of any fish in Elkhorn Slough): "The importance of Shiner Perch in the diets of Caspian and Forster's terns reflects their abundance in the slough." (Baltz et al. 1979, p. 22). Surfperch only accounted for 6 of 34 samples collected in the Lauritzen Channel in 2008.

Furthermore, the most important prey selection factor for terns reported by Baltz et al. (1979) was not fish species but size. Virtually all of the shiner surfperch extracted from Forster's tern stomachs in this study were young of the year juveniles that measured 40 mm standard length or shorter—less than 1.6 inches (see Baltz et al. 1979, Figure 1). According to Baltz et al. (1979), the maximum size prey of any fish species that Forster's terns have ever been observed to take is 75.6 mm standard length (just under 3 inches). Not a single shiner surfperch caught during the 2008 fish sampling survey was in this range. All were larger (3 to 6 inches, with an average length just over four inches; see CH2M Hill 2008, Table 1). Based on the length vs. age information for shiner surfperch reviewed by Baltz et al. (1979), the 2008 fish samples used to develop the surfperch bioaccumulation model were all at least from the year 1 to 2 size classes (75 to 10 mm standard length), and the largest ones were much older. Age is important, because accumulation of hydrophobic chemicals like organochlorine pesticides is a strong function of individual age as well as species. Young of the year surfperch and other fish species small enough for Forster's tern to prey upon likely do occur in the Lauritzen Channel but were excluded from collection by the trawling gear used. However, they likely have far less accumulated DDT in their tissue than the larger, older fish collected. Applying the species-specific bioaccumulation models in the way CH2M Hill (2010a) did likely results in a significant overestimation of tern exposure and risk. In fact, the only fish caught in the Lauritzen Channel that were small enough for Forster's tern to prey on were anchovies (3 composite samples of 43 fish each) and a single goby composite sample of three fish (CH2M Hill 2008, Table 1). The surfperch bioaccumulation model cannot be used to predict exposure of Forster's terns that forage in the Lauritzen Channel. Use of the anchovy model would be far more appropriate. Anchovy is the only species included in the 2008 fish tissue data that is relatively abundant and consistently of the appropriate size class to be representative of Forster's tern prey.

Double-crested cormorant were modeled by CH2M Hill (2010a) to have even higher reliance on shiner surfperch than tern (93% surfperch, 3% goby, and fractional percentages of other species). The cited reference for this assumption is Ainley et al. (1981), which is a study of the dietary preferences of three cormorant species at 18 Pacific coast sites ranging from Alaska to Baja, including one northern California site in the Farallon Islands. At the Farallon site, shiner surfperch accounted for 78.6% of double-crested cormorant diet, and the surfperch family Embiotocidae as a whole accounted for approximately 93% (Ainley et al. 1981, Appendix 3). However, this is likely to reflect local abundance rather than a true preference. Double-crested cormorants are feeding generalists, not specialists. A monograph on wildlife management of the species summarizes dietary selection as follows: "Double-crested cormorants feed almost exclusively on fish, primarily small bottom dwelling or schooling 'forage' fish. They are adaptable, opportunistic feeders that prey on many species of small fish (less than six inches), usually feeding on those that are most abundant and easiest to catch;" furthermore, "Because a cormorant's ability to catch a particular species of fish depends on a number of factors (distribution, relative abundance, behavior, habitat), the composition of a cormorant's diet can vary quite a bit from site to site and throughout the year, and can reflect the number and types of fish present in a given area at a given time" (Sullivan et al. 2006). While the surfperch sampled in the Lauritzen Channel are within the prey size range of cormorants, there is no reason to believe they would be consumed to a degree beyond their proportional abundance. At the other sites in the Ainley et al. (1981) study, where double-crested cormorant data were collected, Embiotocidae accounted for just zero to 21% of the diet. Other fish species which are abundant in the Lauritzen Channel likely make up far higher percentages of the cormorant diet. In particular, anchovies are another favored prey item. The CH2M Hill cormorant model assumes anchovies make up just 0.3% of the diet, which is consistent with the reported data from the Farallon site (Ainley et al. 1981). At two other California sites in the Channel Islands, however, anchovies made up 15 to 23% of the diet, and the most important prey were rockfish (*Sebastes* sp.) and white croaker (*Genyonemus lineatus*), not surfperch (Ibid.). The available fish data for the Lauritzen Channel do not permit a precise description of the forage fish community, but the sample count alone suggests that species other than surfperch may be as important or more important cormorant prey. Given the uncertainties about the CH2M Hill bioaccumulation models in general and the shiner surfperch models in particular, a more representative fish bioaccumulation model should be used for sediment RBC calculation.

5) Area use was not considered.

All of the ecological receptor food web models in CH2M Hill (2010a) assume an area use of 100%, implying that the receptor populations being modeled obtain their entire diet from the Site. This extreme assumption is appropriate only in a screening-level ecological risk assessment, not in a determination of appropriate cleanup targets. No consideration is made for migration periods or for actual forage areas, which are known to be much larger than the Lauritzen Channel for piscivorous birds. The foraging patterns of Forster's terns in particular have been extensively studied in San Francisco Bay using radio tagging and tracking methods over many years. The average daily forage radius for Forster's terns has been reported at 4.9 km from nest sites studied in south San Francisco Bay (Bluso-Demers et al. 2008). Given this range and the extensive habitat present throughout the bay, which is equally suitable or more suitable for tern foraging, the actual area use of the Lauritzen Channel can be expected to be quite small. The ecological risk reassessment contains no discussion of or justification for this important factor, and EPA apparently did not consider it in their selection of sediment RBCs, perhaps because fish RBCs were considered protective of piscivorous wildlife. However, any risk management decision made to protect piscivorous birds should incorporate realistic assumptions about site use.

Section 4.2.1 Human Health RG Reassessment

The draft FFS proposes a human health sediment remediation goal for Total DDT of 450 µg/kg, based on a non-cancer fish tissue risk-based concentration (RBC) of 0.86 mg/kg (wet wt). The draft FFS also states this RBC corresponds to a cancer risk between 10^{-5} and 10^{-4}, within EPA's risk management range. Both the non-cancer and cancer RBCs were derived in the updated human health risk evaluation (CH2M Hill 2010b).

The following equation was used to estimate chemical intake from fish for the purpose of evaluating non-cancer effects:

$$RBC\ (mg/kg) = \frac{THQ \times BW \times AT}{\frac{1}{RfD} \times EF \times ED \times Frac_s \times IR_{fish} \times CF}$$

where,

RBC	=	risk-based concentration in fish (mg/kg, wet wt)
THQ	=	target hazard quotient (1.0 [unitless])
BW	=	body weight (70 kg)
AT	=	averaging time (10,950 days)
RfD	=	oral reference dose for DDT (0.0005 mg/kg/day)
EF	=	exposure frequency (350 days/year)
ED	=	exposure duration (30 years)
$Frac_s$	=	fraction of fish consumed from study area (0.5 [unitless])
IR_{fish}	=	fish consumption rate (85.1 g/day)
CF	=	conversion factor (10^{-3} kg/g)

The human health RBCs assume a fish consumption rate of 85.1 g/day, based on the 95th percentile fish consumption rate from a study conducted among the Laotian community in West Contra Costa County (APEN 1998). Fish consumption rates in this study were derived by combining the results of questions about usual portion size and frequency of eating fish. Although detailed data tables or analysis results are not provided, the study also reports the following: mean fish consumption rate = 18.3 g/day, median = 9.1 g/day, 90th percentile = 42.5 g/day, and 95th percentile = 85.1 g/day.

Flaws in the Fish Consumption Study Design

The APEN (1998) study has several methodological and reporting limitations that make it unsuitable for use in regulatory decision making.

1) Nonspecific Survey Questions

The survey questions relating to portion size and frequency of fish consumption were multiple choice by design and, in many cases, included answer choices that were too broad. For example, information on portion size was elicited by showing a model of a 3-ounce filet and asking respondents how much they typically ate relative to that amount with the possibilities limited to 0.75, 1.5, 3, 4.5, or 6 ounces. Similarly, the possible responses for frequency of fish consumption (of any type) were >1×/day, 1×/day, 3–4×/week, 1–2×/week, few times/month, or <1×/month or never. While this form of question can provide qualitative information on the size and frequency, the possible responses are too nonspecific to allow accurate quantitative information. For example, if two respondents who eat the same portion size each answer that they typically eat fish 1-2x/week, there could be a two-fold difference in their actual frequency of consumption (i.e., once or twice per week). Both would be assigned the same fish consumption rate despite one eating fish at twice the rate of the other. Similarly, a "few" times/month, the most common response, could seemingly include anywhere from 2 to 3 times per month. APEN (1998) did not report how the range of possible values for each answer was reduced to provide a single value per respondent for their analysis.

2) Portion Size Estimates are Highly Uncertain

As described above, typical portion size eaten was elicited by comparison to a model of a 3-oz. filet. However, use of this method is unlikely to provide valid data for this population. As noted in the original study report, many or most in this community typically eat family-style meals, where food is not divided up onto individual plates but rather eaten from one communal platter. Fish, when eaten, is also commonly served in mixed dishes rather than as individual filets. Under these circumstances, the average portion size for a family member could only reasonably be estimated from information about the amount of fish that went into the dish and the number of people eating.

3) Seasonal Differences were Not Incorporated

The fish consumption rate estimates were based on results from questions about typical frequency of consumption in the 4 weeks prior to survey administration. Although the specific dates of survey administration are not clearly reported, the surveys appear to have been administered in the summer months just after survey staff were trained in June 1997. This is important because of large seasonal differences in fish consumption. In fact, as documented in Figure 16 of APEN (1998), most respondents eat fish much more frequently in the spring and summer than in the fall and winter. For example, people most commonly reported fishing between 2–3×/month and >1×/week in the summer but <1×/month in the winter months. Thus, the reported fish consumption rate represents patterns during the highest fish consumption months. If the raw study data were available, seasonal fish consumption rates could be estimated and an overall time-weighted yearly rate estimated.

The Fish Consumption Study is Inappropriately Applied to the Site

In addition to the methodological issues inherent to the APEN (1998) fish consumption study that limit its use for regulatory decision making in general, several factors limit its applicability to the Site.

1) APEN (1998) Represents a Freshwater Fishing Population.

APEN (1998) reports that 77.7% of respondents do not fish in the marine waters of San Francisco Bay. The majority of individuals in this study population fish in lakes, reservoirs, rivers, and delta areas. The freshwater areas of San Pablo Reservoir and Lake Sonoma were the most commonly listed fishing locations and were identified by approximately 50% of respondents as the place they fish most often. Although marine fish are caught and consumed by this population, most fishing occurs in freshwater locations. Freshwater fishing practices cannot be extrapolated to marine fishing populations.

2) Surfperch is Not a Representative Species for Estimating Bioaccumulation

The human health sediment RBCs were derived by applying a sediment-biota regression relationship for surfperch to the fish tissue RBCs based on fish consumption. Surfperch were selected because it provided the most conservative regression relationship. However, use of these data is inconsistent with information about fish consumption patterns in the fish consumption survey selected to be representative of the site. As noted previously, the Laotian community studied in APEN (1998) is primarily a freshwater fishing population, and even among those who fish in marine waters, surfperch is not a particularly popular choice. Only 9 of 95 respondents reported catching surfperch. The most common fish species caught were catfish (n = 45 of 95 respondents), striped bass (n = 41), trout (n = 38), and crappie (n = 35). Striped bass was most frequently reported as the fish most commonly caught by an individual, whereas surfperch was only identified as the most commonly caught fish by one person.

The available data indicate that the site-specific sediment-biota regression model based on all fish would be more appropriate than the surfperch regression model, both because surfperch are not commonly harvested by area anglers and because high fish-consuming populations harvest a wide variety of fish.

3) Inappropriate Use of a High-end Consumption Rate from a High-consuming Population

Policy and public health considerations dictate that health-based limits are typically derived based on consideration of a reasonable maximum exposure (RME) scenario. The RME is designed to represent a high-end (but not worst-case) estimate of individual exposures. The RME is defined as reasonable because it is a product of several factors that are a mix of average and upper-bound estimates (USEPA 1989). By convention, RME estimates typically fall between the 90th and 95th percentile of an exposure distribution. In other words, when all assumptions are taken together, the resulting exposure estimate should be in the range of the 90th and 95th percentile of exposure for the population of concern. Therefore, every individual input (e.g., fish consumption rate, fish diet fraction from the site, exposure duration) should not be at the high end of the distribution in order for the overall exposure estimate to be at the high end of the distribution. For example, the U.S. FDA designates a high-end consumption rate as the 90th percentile from large national, 2 to 3 nonconsecutive day surveys of food intake by thousands of individuals (U.S. FDA 2006).

The specific percentile(s) selected should be considered on a study-specific basis and will depend on such factors as the characteristics of the data distribution and the representativeness of the study population to which the fish consumption rate will be applied. The intent of the RME approach is to ensure protection at the upper end of a distribution that includes the entire population (or in the case of fish consumption, all people who consume fish). The 95th percentile intake from APEN (1998) represents well over the 99th percentile consumption rate for fish consumers among the general public in the U.S. (Polissar et al. 2012), whereas the 90th percentile from APEN (1998) study (42.5 g/day) is similar to the 95th percentile for fish consumers among the general public (43.3 g/day).

The 90th percentile fish consumption rate from APEN (1998) provides a high degree of protection for a high fish consuming population, is highly protective of the general fish consuming population (Polissar et al. 2012), and is consistent with public health protection goals in the U.S. (U.S. FDA 2006). Thus, use of a fish consumption rate of 42.5 g/day for the purpose of risk assessment and to set remediation goals would be highly protective for the site.

4) Fish Fractional Intake from the Site is Drastically Overstated

The fish tissue RBC calculations assume that 50% of fish consumed comes from the site ($Frac_s = 0.5$). This assumption is based, in part, on information reported in APEN (1998). CH2M Hill (2010b) states that "the APEN study found that 42.8 percent of the survey respondents had eaten fish caught from locations other than the San Francisco Bay in the past 4 weeks and that 55.9 percent had eaten fish from a store or restaurant in the past 4 weeks." Although this is consistent with the information reported in APEN (1998), the APEN study was conducted in the summer, a time of year when fishing frequency is at its highest level. As discussed previously, fishing frequency is much higher in the spring and summer than in the fall and winter. Fractional intake from the bay is thus likely to be much lower in the fall and winter than reported in APEN (1998).

The frequency of fishing from the San Francisco Bay is not the same as fishing from one small waterway like the Lauritzen Channel. A fractional intake of 0.5 from the site is highly unlikely because both the area and fish resource are too small to sustain half the intake of a high end fishing population over 30 or more years, and because industrial activities would make it difficult to fish the site at anywhere near the frequency needed to reach this usage rate. The Lauritzen Canal does not have any piers, beaches, or other shoreline amenable for fishing. There are also several state of California fish advisory signs posted around the Channel. Finally, the Lauritzen Channel is a secured location designed to prevent this very exposure. Because the Channel is an active marine terminal it is subject to homeland security requirements and the entire area is fenced in. On the other hand, there are more appealing public fishing areas within close proximity, including nearby Marina Bay and Point Richmond.

The fish fraction from the site assumption should represent a more realistic, but still conservative, health-protective scenario. Little or no fishing is likely to occur in the Lauritzen Channel under current use. However, even if site-use conditions were to change in the future, the fractional use of the site, particularly by a high fish-consuming population, would likely be very low because people would fish from a wide variety of locations, as demonstrated in the APEN (1998) study. A more reasonable assumption would incorporate the amount of shoreline in the Lauritzen Channel relative to the water body in which it is contained. The shoreline of the Lauritzen Channel represents less than 5% of the Richmond Inner Harbor (from Ferry Point and Point Isabel, including Richmond Marina Bay and Santa Fe Channel). The relative surface area would be much smaller. The small area, in combination with the preference of area anglers for fishing locations outside the Richmond Inner Harbor and the high percentage use of store and restaurant purchased fish, indicates fractional use of the site would be even lower, likely less than 1%. Therefore, for the purpose of risk assessment and development of remediation goals, a fish fraction assumption of 0.1 (i.e., 10%) would be highly protective for the site.

Chapter 5 Identification and Screening of Remedial Technologies

This short chapter purports to lay out the criteria by which remedial options were screened and selected for subsequent development into specific alternatives. However, the discussion and justification for rejecting all available technologies beyond dredging is weak to nonexistent. Little rationale is provided for scoring of rejected alternatives, and, in some cases, the scores appear inconsistent with other information in the FFS. In particular, *in situ* treatment technologies, including activated carbon amendment, were given low scores for effectiveness (FFS Table 5-3), despite being described elsewhere in the report as effective and promising, with a 90 to 99% reduction in apparent bioavailability of DDT (see FFS, Section 2.8). Further, EPA provides no justification for why it only retained carbon amendment for further evaluation in a limited capacity in two of the proposed remedial alternatives as a source control measure. In addition, MNR and ENR are dismissed due to “site conditions” with little further explanation other than a reference to the flawed STS.

Chapter 6 Development and Analysis of Remedial Alternatives

The scope of specific remedial alternatives developed in the draft FFS is extremely narrow. Beyond the no action alternative (included only as a benchmark of zero effectiveness), three options are assessed. All rely on dredging most of the Lauritzen Channel with limited use of an active cap at the upper end of the channel as a source control measure for potential (albeit poorly characterized) groundwater and stormwater inputs. The dredge footprint areas (7 to 8.4 acres, or about 74 to 88% of the channel) and the estimated costs ($21.7 million to $22.7 million) are virtually identical. Only slight changes in the footprint account for the differences between alternatives. Beyond the flawed risk analysis used to develop the amended RGs and the inadequate assessment and justification for candidate remedial technologies noted above, the approach taken by EPA suffers from a lack of appropriate spatial evaluation of contamination and remedial benefit (i.e., the spatial distribution of sediment contamination is never quantitatively factored into the remedy selection process). By failing to quantitatively link the extent of proposed remediation with exposure reduction, EPA is effectively mandating a large-scale cleanup without quantitative justification. In fact, much more limited and cost-effective cleanup options could effectively protect beneficial uses, especially if risk-based RGs were reassessed using reasonable and scientifically valid exposure and toxicity assumptions.

Remedial Goal Revision

The extent of remediation is ultimately driven by the risk-based RGs. In the draft FFS, these have been set using flawed and unrealistic risk evaluations, (see discussion above). In order to support a reasonable cleanup selection, appropriate risk-based sediment targets must first be derived. The first step in deriving protective sediment RBCs is to calculate reasonable, protective tissue RBCs.

Appropriate Ecological Fish Tissue RGs

<u>Risk to Fish</u>

The ecological risk-based RG for DDT selected in the draft FFS is 400 µg/kg total DDT in sediment. This value is based on a sediment RBC for shiner surfperch developed in the 2010 ecological risk reassessment that is ultimately driven by a fish tissue residue effect threshold estimate of 0.60 mg/kg (wet wt). Correction of the flaws noted above in the interpretation of the source compilation (Beckvar et al. 2005) and use of a more representative and technically defensible adverse effect threshold estimate for fish results in a much higher exposure threshold for shiner surfperch and all other fish.

Given the limitations of the Beckvar et al. (2005) compilation of fish tissue residues reported to be associated with adverse effects, a more appropriate tissue-based threshold for DDT would a value somewhere in the range of LER endpoints (see Beckvar et al. 2005, Table 3). Excluding the goldfish behavioral study (which reported no population-level, ecologically relevant endpoints), the eight remaining values range from 0.55 to 112.7 mg/kg. A reasonably low-biased central tendency of these data would be the geometric mean of the eight LERs. Such a concentration would at least be associated with a significant probability of adverse effects in the tested organisms and would capture the empirical variability of the diverse source studies. This value is 4.62 mg/kg (wet wt), which is 7.7-fold higher than the t-TEC from Beckvar et al. (2005) and far more reasonable for the purposes of setting risk management goals. This value should be considered protective of all fish species in the Lauritzen Channel.

<u>Risk to Piscivorous Birds</u>

As noted above, the DDT risk driver for fish-eating wildlife is the Forster's tern. The same LOAEL TRV was used to estimate dietary effect thresholds for all three modeled bird species, but this small bird (149 g average body wt) has a lower body-weight-adjusted daily dose threshold than larger birds. Using the ingestion rate (90 g/day), body weight, and TRV for DDT (281 µg/kg bw/day) selected by CH2M Hill (2010a, Table 16), the predicted threshold fish tissue concentration may be calculated as follows:

$$[Fish\ Tissue] = \frac{TRV\ \times body\ wt}{Ingestion\ rate} = \frac{\frac{281\ \mu g\ DDT}{kg\ bw\ day} \times 0.190\ kg\ bw}{0.090\ kg/day} = 593\,\frac{\mu g}{kg}\,(wet\ wt)$$

This value will also be protective of all larger piscivorous birds. The predicted fish tissue RBC for female cormorant would be calculated as follows:

$$[Fish\ Tissue] = \frac{TRV\ \times body\ wt}{Ingestion\ rate} = \frac{\frac{281\ \mu g\ DDT}{kg\ bw\ day} \times 1.831\ kg\ bw}{0.583\ kg/day} = 882\,\frac{\mu g}{kg}\,(wet\ wt)$$

Appropriate Human Health Tissue RGs

Table 3 summarizes assumptions used to derive the fish tissue DDT RBC by CH2M Hill (2010b) alongside an alternative approach using site-specific and more realistic, but highly health-protective, assumptions as described in the comments above.

The alternative risk-based concentration uses a 90th percentile fish consumption rate from APEN (1998) and fish fraction from the site of 10%. Both RBCs are based on non-cancer health endpoints, and, as with the RBC derived by CH2M Hill (2010b), the alternative RBC corresponds to a cancer risk between 10^{-5} and 10^{-4}, within EPA's risk management range. The resulting tissue RBC is 8.59 mg/kg (wet wt) in edible tissue, a value 10-fold higher than the overly-conservative value calculated by the flawed assessment of CH2M Hill (2010b).

Sediment RBCs Derived using Appropriate Bioaccumulation Models

In order to calculate DDT sediment RBCs that are protective of fish, anglers, and piscivorous wildlife, a scientifically valid and representative DDT bioaccumulation model for fish must be selected. The model should be site-specific and should be predictive over the entire range of expected post-remedial sediment and tissue concentrations. Species-specific models should be used when available for fish. Bioaccumulation models used to calculate RBCs for humans and piscivorous wildlife must be representative of the fish species that are actually consumed in order to reflect realistic exposure conditions.

Shiner Surfperch Sediment RBC

As discussed above, the surfperch bioaccumulation models developed by CH2M Hill (2010a) are flawed and should not be considered reliable for prediction of post-remedial pesticide uptake. However, the limitations of the available fish tissue data (i.e., the inability to associate individual fish tissue samples from the Lauritzen Channel with any specific sediment concentration) make it difficult to generate a more reliable model from these data. For the purposes of setting a DDT sediment RBC for protection of shiner surfperch (and therefore all other species of fish with lower DDT accumulation rates), we can apply the existing logistic regression surfperch models to the revised surfperch tissue RBC of 4.62 mg/kg (wet wt).

The shiner surfperch bulk concentration bioaccumulation model is as follows (CH2M Hill 2010a, Table 9):

$$\ln[Fish\ DDT\ (\mu g/kg\ wet\ wt)] = 2.667 + 0.668\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Solving this equation for sediment concentration yields a dry wt sediment RBC of 5,650 µg/kg, a value 14 times higher than the mean RBC from the 2010 ecological risk reassessment (CH2M Hill 2010a, Table 21), which was also the RG proposed in the draft FFS to protect surfperch.

The lipid/TOC normalized bioaccumulation model for shiner surfperch is as follows (CH2M Hill 2010a, Table 10):

$$\ln[Fish\ DDT\ (\mu g/kg\ lipid)] = 3.6023 + 0.5865\ \times \ln[Sediment\ DDT(\mu g/kg\ dry\ wt)]$$

Applying the average surfperch lipid level measured in the Lauritzen Channel (4.05%, n = 6 fish), the lipid-normalized surfperch tissue RBC would be 140,000 µg/kg lipid. Solving for the predicted sediment concentration yields a TOC normalized sediment RBC of 1,270,000 µg/kg TOC. Conversion of this value to dry weight, using the average TOC level measured in the Lauritzen Channel (2.2%, n = 10 fish), yields a dry weight sediment RBC of 27,900 µg/kg. This value is approximately 70 times higher than the mean 2010 reassessment RBC. Based on this reanalysis, it seems clear that shiner surfperch and resident fish in general should not be risk drivers at this site.

Piscivorous Bird RBCs

As noted above, the 2008 surfperch data from CH2M Hill (2010a) are fundamentally inappropriate for modeling exposure to Forster’s tern because they are of the wrong size class. Forster’s terns do not eat these larger surfperch. Anchovy is the only fish species for which data exist in the appropriate size class, and the only bioaccumulation models developed in the 2010 ecological risk reassessment that are appropriate to model tern exposure are those based on the anchovy data. The anchovy logistic regressions, like all bioaccumulation models based on the 2008 fish tissue data, are of questionable reliability because of the unknown sediment concentration associated with individual samples from the Lauritzen Channel. However, application of the regressions to the Forster’s tern tissue RBC of 593 µg/kg yields the following.

The anchovy bulk concentration bioaccumulation model is as follows (CH2M Hill 2010a, Table 9):

$$\ln[Fish\ DDT\ (\mu g/kg\ wet\ wt)] = 2.400 + 0.475\ \times \ln[Sediment\ DDT(\mu g/kg\ dry\ wt)]$$

Solving this equation for sediment concentration yields a dry weight sediment RBC for tern of 4,400 µg/kg, a value 10 times higher than the mean sediment RBC for protection of Forster’s tern in the 2010 ecological risk reassessment (440 µg/kg; CH2M Hill 2010a, Table 21).

The lipid/TOC normalized bioaccumulation model for anchovy is as follows (CH2M Hill 2010a, Table 10):

$$\ln[Fish\ DDT\ (\mu g/kg\ lipid)] = 4.3934 + 0.4404\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Applying the average anchovy lipid level measured in the Lauritzen Channel (1.67%, n = 3 composites of 43 fish each), the lipid-normalized tissue RBC for tern would be 35,500 µg/kg lipid. Solving for the predicted sediment concentration yields a TOC normalized sediment RBC of 999,000 µg/kg TOC. Conversion of this value to dry wt, using the average TOC level measured in the Lauritzen Channel (2.2%, n = 10) yields a dry weight sediment RBC of 22,000 µg/kg. This value is approximately 50 times higher than the mean sediment RBC for protection of tern in the 2010 risk reassessment.

The cormorant fish tissue DDT RBC is 49% higher than the tern RBC (882 and 593 µg/kg wet wt respectively). Because of the non-linear relationship between fish tissue and sediment concentrations, this ratio is not fully proportional when translated to sediment RBCs. However, the tern values are protective of cormorant when the same fish bioaccumulation models are used. Given the diverse diet of cormorant, the logistic regression for all fish is more realistic than any single-species regression. The bulk concentration bioaccumulation model for all fish combined is as follows (CH2M Hill 2010a, Table 9):

$$\ln[Fish\ DDT\ (\mu g/kg\ wet\ wt)] = 2.320 + 0.575\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Solving this equation for sediment concentration yields a dry weight sediment RBC for cormorant of 2,345 µg/kg, a value more than 3 times higher than the mean female cormorant RBC of 700 µg/kg from the 2010 ecological risk reassessment (CH2M Hill 2010a).

The lipid/TOC normalized bioaccumulation model for all fish is as follows (CH2M Hill 2010a, Table 10):

$$\ln[Fish\ DDT\ (\mu g/kg\ lipid)] = 4.9732 + 0.4468\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Applying the average lipid level measured in all Lauritzen Channel fish (2.30%, n = 34 samples), the lipid-normalized fish tissue RBC would be 38,300 µg/kg lipid. Solving for the predicted sediment concentration yields a TOC normalized sediment RBC of 266,000 µg/kg TOC. Conversion of this value to dry weight, using the average TOC level measured in the Lauritzen Channel (2.2%, n = 10 samples) yields a dry weight sediment RBC of 5,854 µg/kg. This value is more than 8 times higher than the mean 2010 reassessment RBC for female cormorant.

Human Health RBC

The 2010 human health risk reassessment (CH2M Hill 2010b) used bioaccumulation models for shiner surfperch to predict a sediment RBC that would be protective of human health, a decision that was used without review by the draft FFS. Apart from the problems associated with the surfperch bioaccumulation model that are reviewed above, the exclusive use of surfperch data to predict human exposure is inappropriate and scientifically unjustifiable. The implied assumption that surfperch are a highly consumed species relative to all other sampled fish is not justified by the 2010 risk reassessment or the FFS and is contradicted by the angler survey (APEN1998) relied upon by CH2M Hill (2010b) to quantify human fish ingestion. In fact, there is no basis to select any single bioaccumulation model for calculation of a sediment RBC to protect human health, since humans consume a variety of fish species. Human exposure should be modeled using relationships developed for multiple species of fish.

For the purposes of estimating sediment RBCs from the corrected human health fish tissue RBC of 8.59 mg/kg (wet wt), we have applied two multi-species logistic regressions from the 2010 risk reassessment: all fish and benthic fish.

The bulk concentration bioaccumulation model for all fish combined is as follows (CH2M Hill 2010a, Table 9):

$$\ln[Fish\ DDT\ (\mu g/kg\ wet\ wt)] = 2.320 + 0.575\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Solving this equation for sediment concentration yields a dry weight sediment RBC of 123,000 µg/kg, a value 273 times higher than the human health RBC of 450 µg/kg from the 2010 human health risk reassessment (CH2M Hill 2010b), which was also the RG proposed in the draft FFS to protect human health.

The lipid/TOC normalized bioaccumulation model for all fish is as follows (CH2M Hill 2010a, Table 10):

$$\ln[Fish\ DDT\ (\mu g/kg\ lipid)] = 4.9732 + 0.4468\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Applying the average lipid level measured in all Lauritzen Channel fish (2.30%, n = 34 samples), the lipid-normalized fish tissue RBC would be 373,000 µg/kg lipid. Solving for the predicted sediment concentration yields a TOC normalized sediment RBC of 43,400,000 µg/kg TOC. Conversion of this value to dry weight, using the average TOC level measured in the Lauritzen Channel (2.2%, n = 10 samples) yields a dry weight sediment RBC of 955,000 µg/kg. This value is more than 2000 times higher than the mean 2010 reassessment RBC.

Alternatively, the logistic regressions for benthic fish can be used to calculate a sediment RBC. These samples include fish closely associated with sediments, as well as some larger fish with relatively high bioaccumulation levels, and include some minor game species like starry flounder.

The bulk concentration bioaccumulation model for benthic fish is as follows (CH2M Hill 2010a, Table 9):

$$\ln[Fish\ DDT\ (\mu g/kg\ wet\ wt)] = 2.191 + 0.656\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Solving this equation for sediment concentration yields a dry weight sediment RBC of 35,200 µg/kg, a value 78 times higher than the human health RBC of 450 µg/kg proposed in the draft FFS.

The lipid/TOC normalized bioaccumulation model for benthic fish is as follows (CH2M Hill 2010a, Table 10):

$$\ln[Fish\ DDT\ (\mu g/kg\ lipid)] = 4.7231 + 0.5332\ \times \ln[Sediment\ DDT\ (\mu g/kg\ dry\ wt)]$$

Applying the average lipid level measured in all Lauritzen Channel benthic fish (1.49%, n = 8 samples), the lipid-normalized fish tissue RBC would be 577,000 µg/kg lipid. Solving for the predicted sediment concentration yields a TOC-normalized sediment RBC of 9,000,000 µg/kg TOC. Conversion of this value to dry weight, using the average TOC level measured in the Lauritzen Channel (2.2%, n = 10 samples) yields a dry weight sediment RBC of 199,000 µg/kg. This value is 440 times higher than the human health RBC proposed in the draft FFS.

Summary of Corrected Tissue and Sediment RBCs

The results of the RBC recalculations described above are summarized in Table 4. The RBCs derived in the 2010 risk reassessment memoranda consistently, and in some cases egregiously, exaggerate realistic exposure and risk levels. EPA has apparently not reconsidered the appropriateness of these values or the methods used to derive them in the context of remedial decision-making. Based on this re-evaluation, a significant critical reconsideration should be part of the final FFS. Without alteration, the 2010 RBCs are unsupportable as RGs. When realistic and scientifically justifiable (i.e., RME) assumptions are substituted for the worst-case assumptions used in the 2010 risk reassessment, none of the sediment RBCs for DDT exceeds the original RG from the 1994 ROD (590 µg/kg).

Piscivorous birds (not fish) appear to be the ecological risk driver for DDT based on the estimated sediment RBCs in Table 4, but use of these values to set RGs must still consider the area use question. In their current form, these RBCs assume 100 percent area use, a value that is without question unrealistic. Actual area use can be highly site-specific and difficult to determine. However, extensive data exist on nesting sites for water fowl in the San Francisco Bay area, in particular Forster's terns. Prior to setting any revised cleanup levels to protect birds from DDT exposure, a defensible site-specific area use factor should be developed.

The DDT non-cancer risk sediment RBCs in Table 4 are many multiples higher than those calculated by CH2M Hill (2010b) due to the substitution of RME assumptions for extreme, worst-case assumptions. Determining appropriate levels of risk tolerance is ultimately a policy decision as well as a science decision, but the extreme range of values suggests that EPA should thoroughly re-evaluate the human exposure scenarios and input assumptions before finalizing any amended RGs for DDT at this site.

Example Cleanup Alternative Using Recalculated RGs

A full revised remedy selection is beyond the scope of this review. However, it is possible to demonstrate the magnitude of cleanup that more realistic risk-based RGs for DDT, such as those derived above, would support. The following example is provided for comparative purposes and is not intended to represent a fully optimized remedial design. It does, however, illustrate an approach that could be used to generate protective remedial alternatives that are cost-effective and quantitatively linked to exposure reduction. The points made in the discussion above regarding the incomplete understanding of sources, source control, and sediment fate and transport should still be addressed prior to finalizing the FFS or selecting a remedy.

The goal of any risk-based cleanup should be to reduce area weighted average exposure for the entire channel (i.e., surficial sediment SWAC) to a level that meets all selected RBCs. Figure 2 is a Thiessen polygon map of the Lauritzen Channel generated using the most current surface sediment data from 2013. The current total DDT SWAC for the entire Channel is 7,627 µg/kg. We have evaluated three remedial scenarios with different target cleanup levels. All scenarios consider a combination of hotspot dredging and activated carbon amendment. All polygons with total DDT concentrations above 30,000 µg/kg are included in the dredging footprint for all scenarios, which is approximately 0.9 acres in size. For these calculations, post-remedial total DDT concentrations in dredged areas are assumed to be 66 µg/kg (the current mean value of Santa Fe Channel YBM sediment, see FFS, Table 3-3). Post remedial DDT concentrations in areas treated with activated carbon would be dependent on the mixing and binding efficiency of the amendment used. Because there is uncertainty about the net effectiveness of a carbon amendment remedy, we have modeled two values – 95% exposure reduction (a level demonstrated to be feasible in bench scale testing) and 80% exposure reduction (a value that allows for possible inefficiencies in field-scale implementation). All of the scenarios described below are based on a simple "hill-topping" approach, whereby the highest concentration polygons are remediated first. For a given scenario, the dredging footprint is implemented, then polygons are added to the carbon treatment footprint, in decreasing order of total DDT concentration, until the target SWAC for the Channel is reached.

<u>Scenario 1—Target SWAC = 1,000 µg/kg</u>

The lowest recalculated sediment RBC is 2,345 µg/kg, a value calculated to protect double-crested cormorant assuming 100 percent area use in the Lauritzen Channel (see Table 4). Given the assumptions of the RBC recalculation described above, this RBC should easily be protective of all modeled human and ecological receptors. Cormorant, tern and other piscivorous waterfowl that may use Lauritzen Channel have very large foraging ranges. The small size of the Channel (less than 10 acres) is neither a significant fraction of cormorant foraging range, nor is it a significant fraction of the available local habitat for waterfowl. Accurately estimating area use for wildlife or fractional intake for human receptors is challenging at any site, and conservative approaches are typically used (e.g., RME scenarios, as described above in the human health RBC discussion). However, setting area use factors for wildlife or fractional intakes for humans at 100% at this small, restricted access site yields a worst-case bounding scenario with no relevance to actual exposure of any receptor population. Adjustment of area use / fractional intake to a realistic value would result in a proportional decrease in predicted exposure and increase in sediment RBC. For example, if a 50% area use factor were assumed for cormorant (still a highly conservative value), the predicted sediment RBC would be doubled (i.e., 4,690 µg/kg). However, in the interests of evaluating the magnitude of a protective cleanup that includes a significant safety factor on top of RME assumptions, a SWAC target value of 1,000 µg/kg has been chosen for this scenario. The objective is to demonstrate that a remedy can be designed using hotspot dredging and carbon amendment over a limited area that lowers exposure enough to protect all beneficial uses. Figure 3 illustrates the protective remedy assuming carbon amendment would reduce exposure by 95%. The activated carbon amendment footprint is approximately 0.8 acres in size. Figure 4 illustrates the protective remedy if carbon amendment efficiency is reduced to 80%, resulting in a slightly increased carbon treatment footprint of 1.1 acres. At this target SWAC level, the cleanup footprint is not very sensitive to carbon performance.

Scenario 2—Target SWAC = 590 µg/kg

If the total DDT target SWAC was set at the level stipulated in the 1994 ROD, 590 µg/kg, a larger remedial footprint would be required. This level is clearly below any level required to protect beneficial uses, if realistic exposure assumptions for human and ecological receptors are made. However, the hotspot dredging plus carbon amendment technology discussed above can easily accommodate even this overly-protective cleanup target. Figure 5 illustrates a protective remedy assuming 95% exposure reduction for the carbon treatment with the same dredging footprint modeled under scenario 1. Figure 6 is the hill-topping remedy that would be required if 80% carbon treatment efficiency is assumed. The areas of the carbon treatment footprint for these remedial options are 1.8 acres and 2.9 acres, respectively. The influence of carbon treatment efficiency is proportionally larger at this target SWAC level, but the range is still relatively modest.

Scenario 3—Target SWAC = 400 µg/kg

We have also evaluated the total DDT cleanup target of 400 µg/kg proposed in the draft FFS. While not justifiable from a risk perspective (see above discussion), even this cleanup target is achievable using the hotspot dredging and activated carbon remedial approach we have described. At 95% carbon efficiency, the activated carbon footprint is 2.6 acres in size. At 80% carbon efficiency, the required carbon footprint is more than twice as large at 6.9 acres. While not a realistic assessment of the remedial scope required for protection of beneficial uses, this scenario makes two important points:

- Reliance on carbon amendment as the primary remediation technology can achieve even unrealistically protective remedial goals using a more cost-effective and less disruptive alternative to dredging alone.
- When cleanup targets are derived using inappropriate assumptions about exposure and risk, the result can be remedial designs that are disproportionately high relative to realistic exposure assumptions.

Other remedial footprints and options could achieve the same target SWACs and may be ultimately more cost-effective as the result of engineering feasibility or other considerations, but this exercise provides proof of concept for the combination dredging and activated carbon option as well as a simple sensitivity analysis for the selected sediment RBC.

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Figures

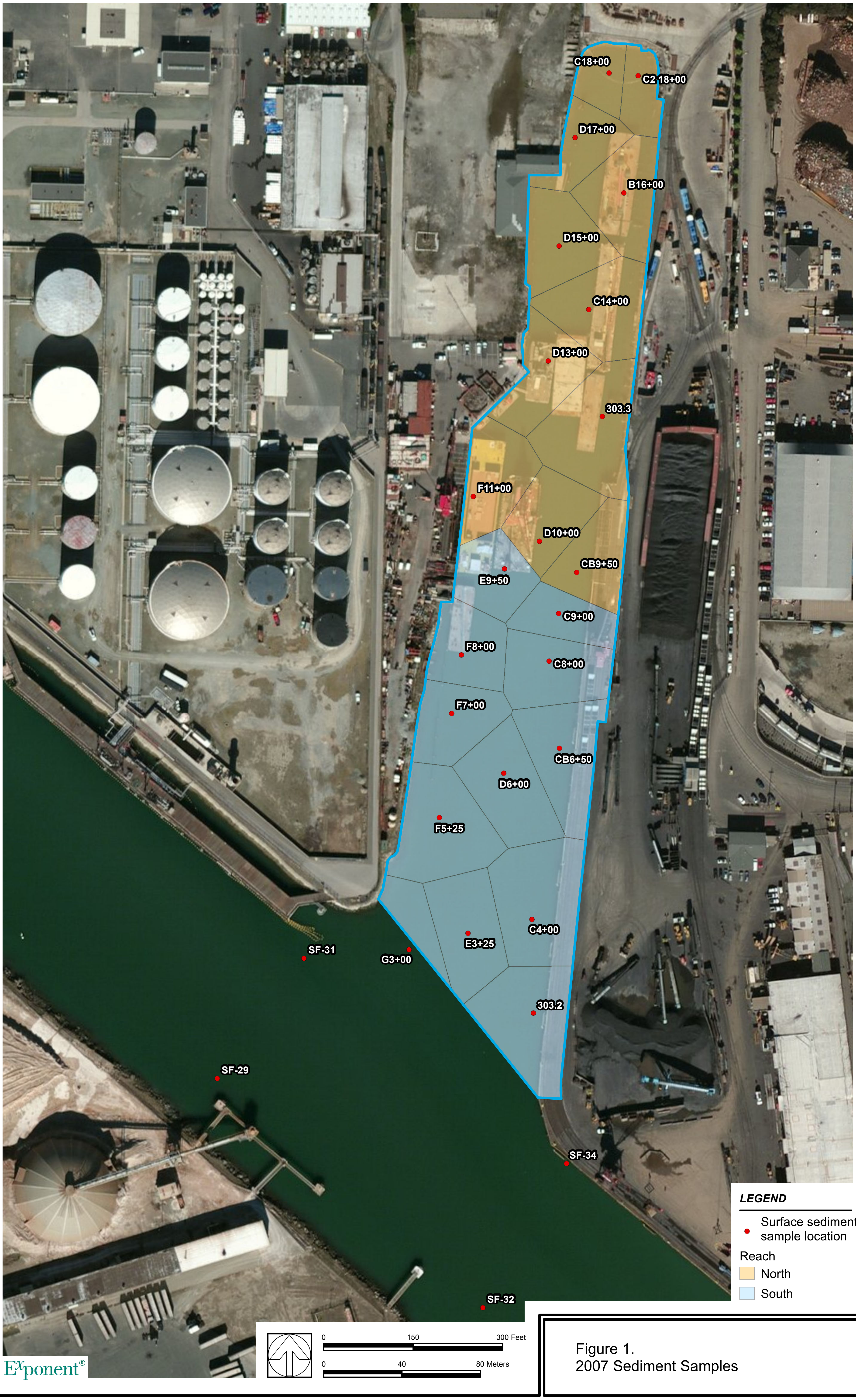

Figure 1.
2007 Sediment Samples

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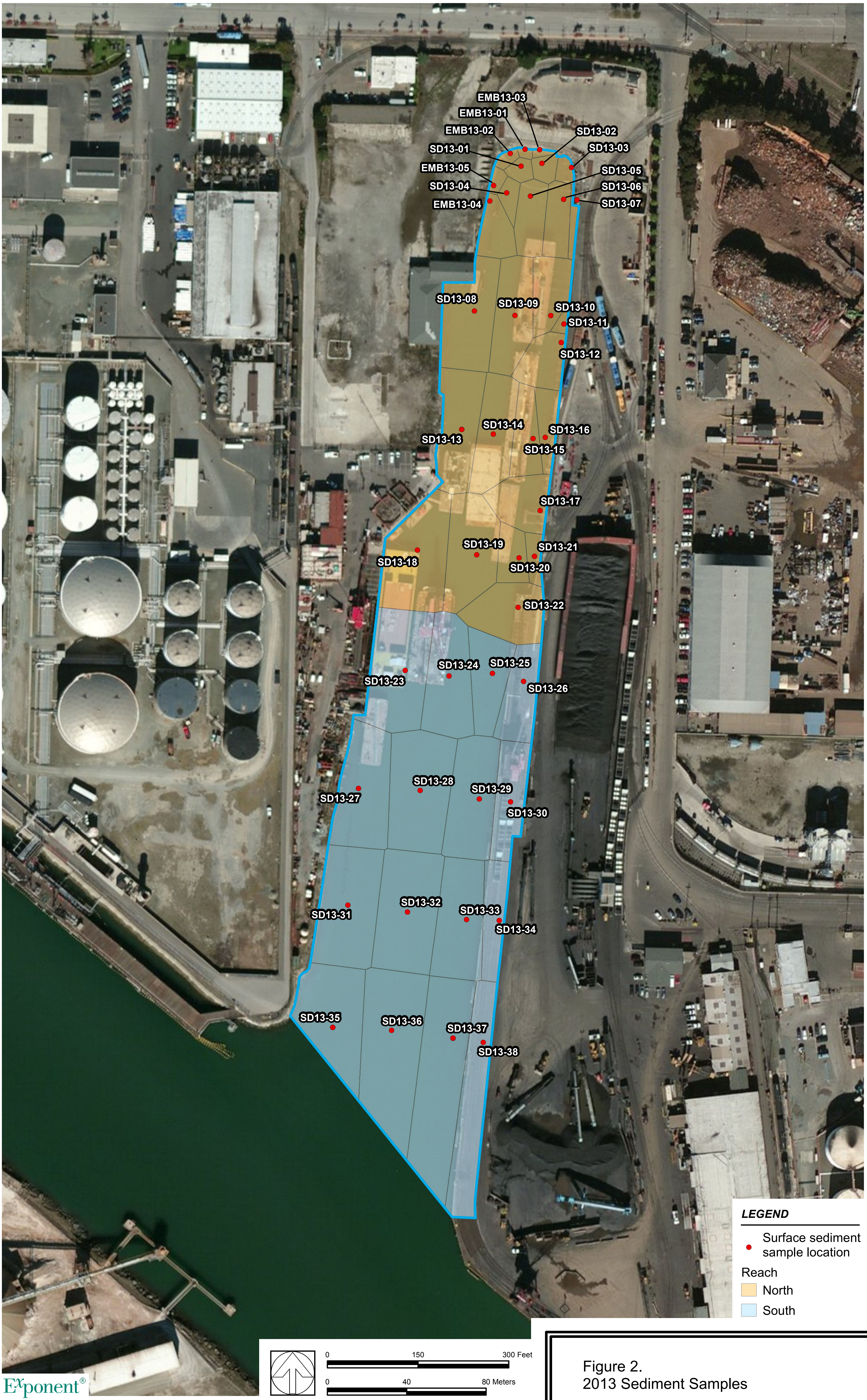

Figure 2.
2013 Sediment Samples

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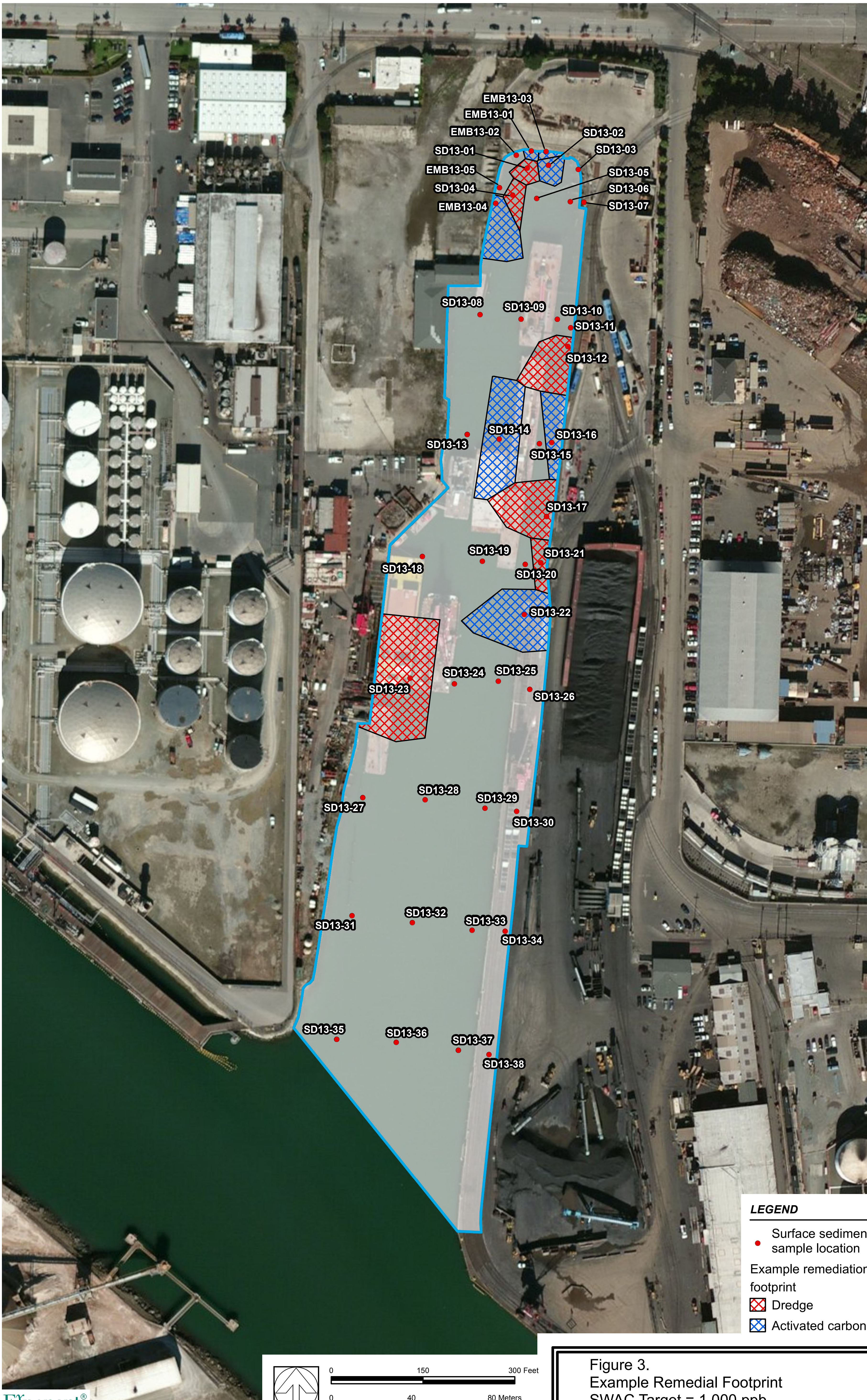

Figure 3.
Example Remedial Footprint
SWAC Target = 1,000 ppb
Carbon Efficiency = 95%

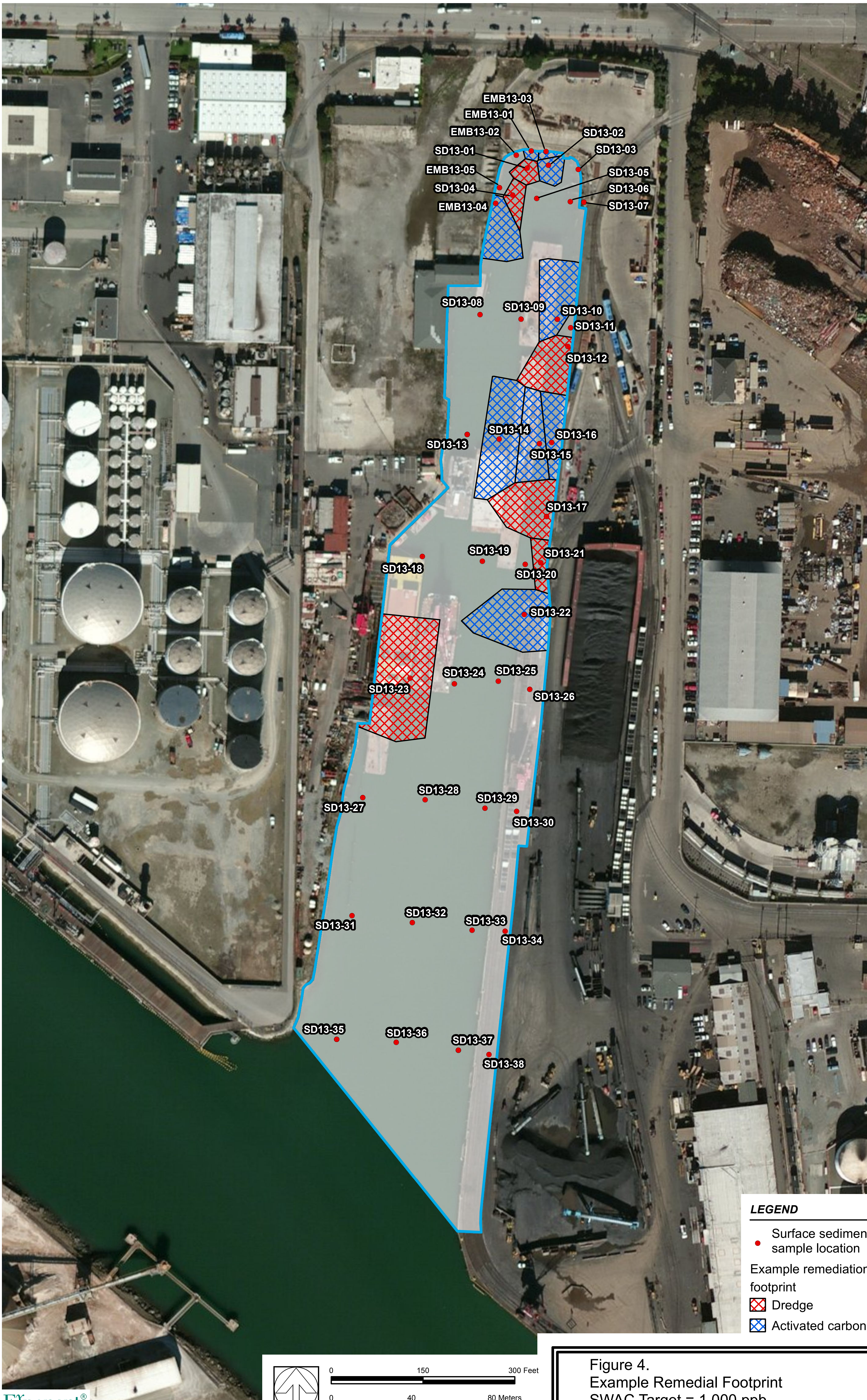

Figure 4.
Example Remedial Footprint
SWAC Target = 1,000 ppb
Carbon Efficiency = 80%

Figure 5.
Example Remedial Footprint
SWAC Target = 590 ppb
Carbon Efficiency = 95%

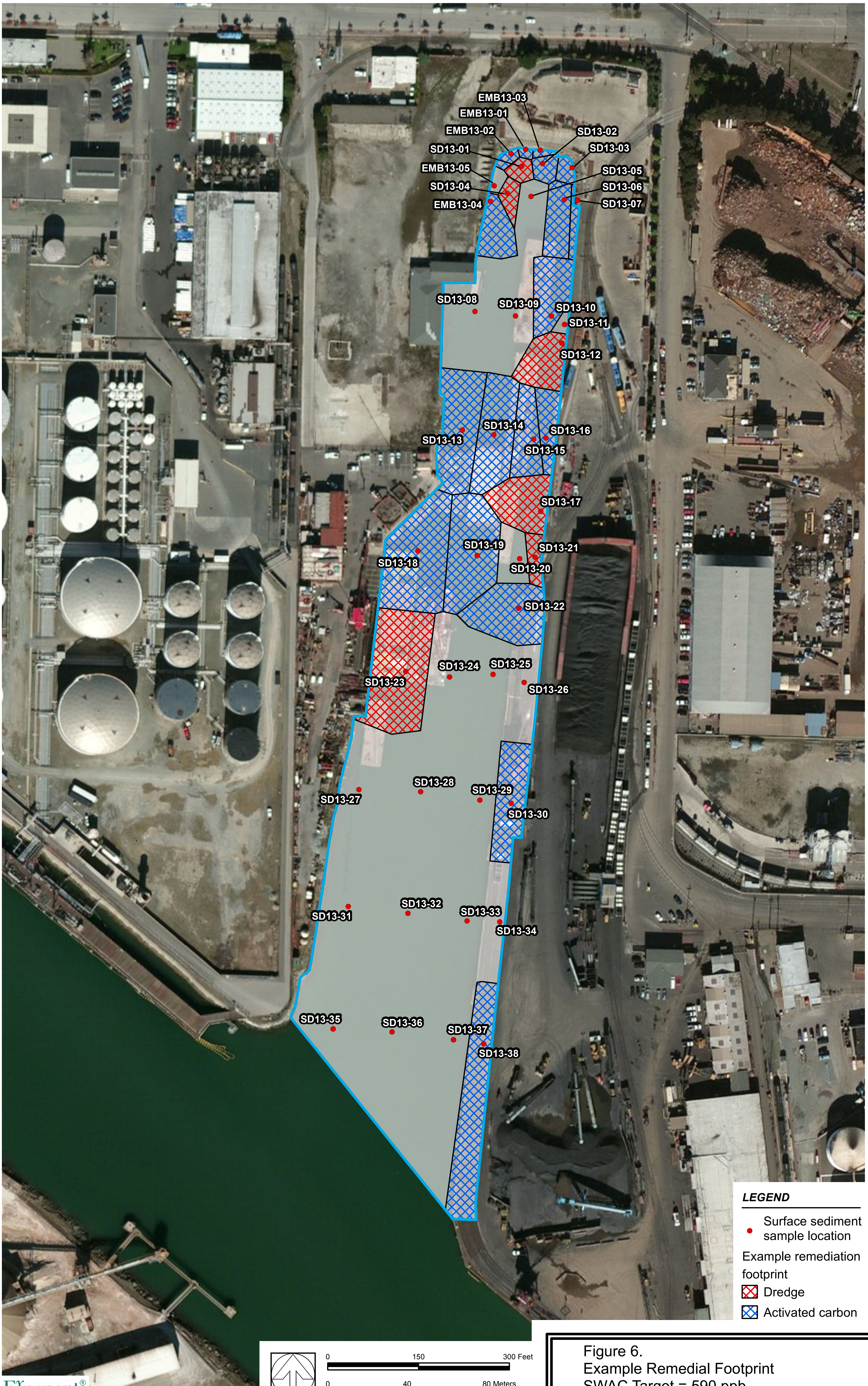

Figure 6.
Example Remedial Footprint
SWAC Target = 590 ppb
Carbon Efficiency = 80%

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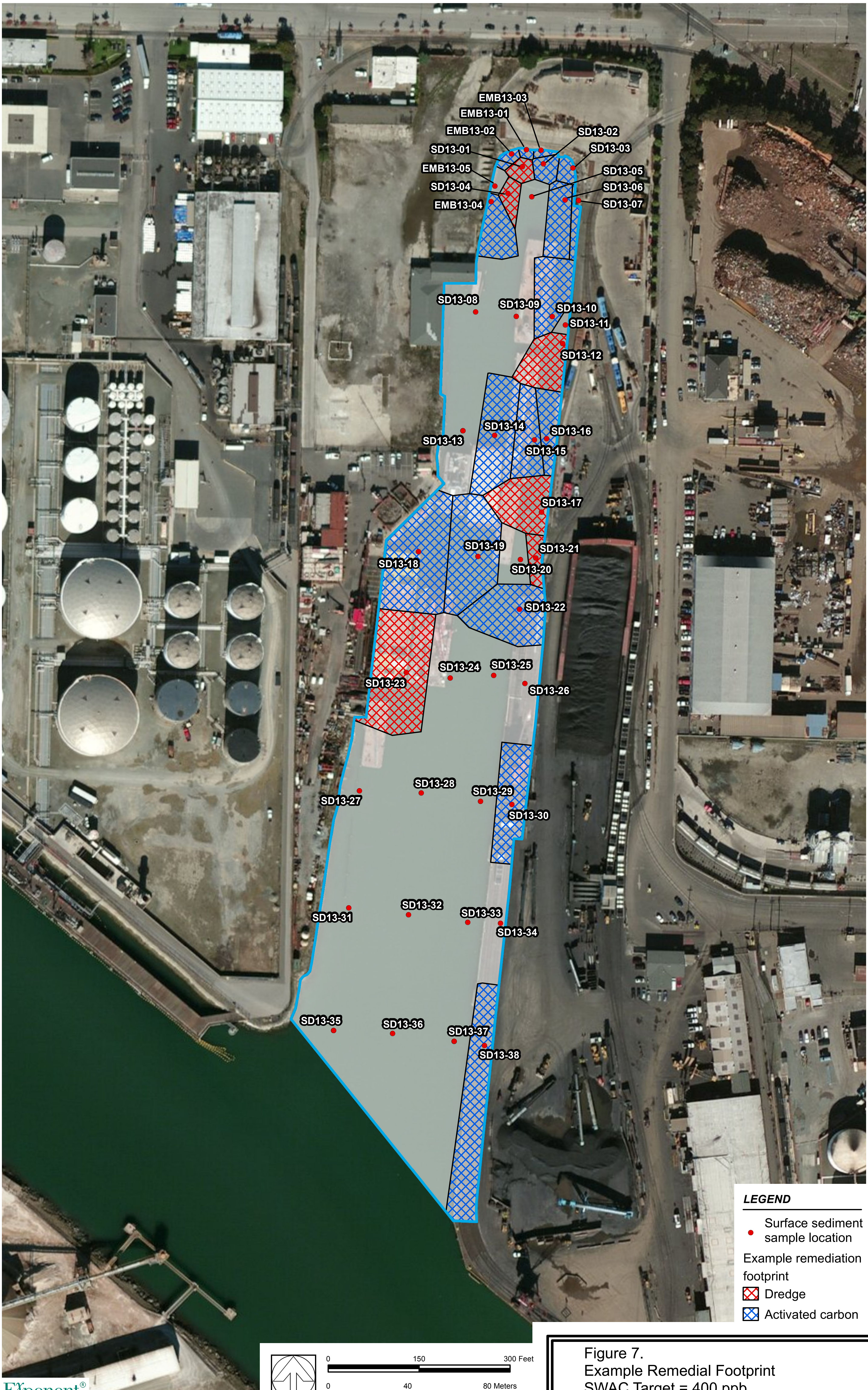

Figure 7.
Example Remedial Footprint
SWAC Target = 400 ppb
Carbon Efficiency = 95%

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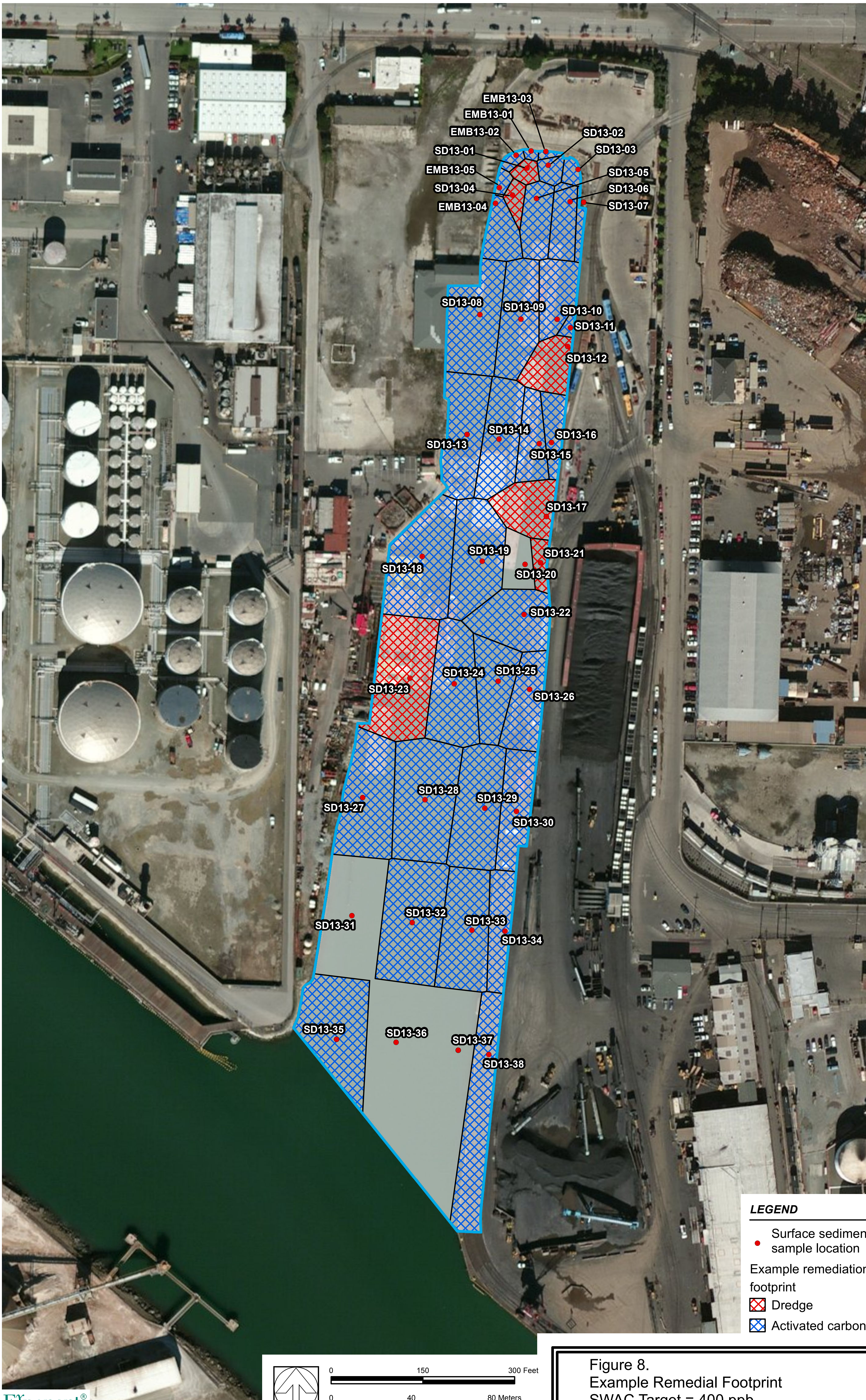

Figure 8.
Example Remedial Footprint
SWAC Target = 400 ppb
Carbon Efficiency = 80%

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Tables

Table 1. Sediment SWAC and mean fish tissue concentrations in the Lauritzen Channel

	Sediments				All Fish and Shrimp			Shiner Surfperch			Benthic Fish		
Channel Reach	Area (ft^2)	Sample Count	DDT SWAC (µg/kg dry wt)	DDT SWAC (µg/kg TOC)	Sample Count	Mean Tissue DDT (µg/kg wet wt)	Mean Tissue DDT (µg/kg lipid)	Sample Count	Mean Tissue DDT (µg/kg wet wt)	Mean Tissue DDT (µg/kg lipid)	Sample Count	Mean Tissue DDT (µg/kg wet wt)	Mean Tissue DDT (µg/kg lipid)
North	183,916	11	12,729	733,646	34	1,888	80,788	6	5,342	122,403	8	2,656	178,160
South	231,295	12	2,492	112,624									
Total	415,211	23	7,026	387,704									

Notes: See Figure 1 for sediment sample locations and reach boundaries.
TOC-normalized SWAC calculated using measured TOC values when available and the Lauritzen Channel average value (2.2%) when not available.
Benthic fish include flatfish (halibut, sanddab, starry flounder), goby, and staghorn sculpin.

Table 2. Range of Possible DDT BSAF Values in the Lauritzen Channel

Sediment Concentration			All Biota				Shiner Surfperch				Benthic Fish			
			Measured BSAF		Regression Model		Measured BSAF		Regression Model		Measured BSAF		Regression Model	
	µg/kg dry wt	µg/kg TOC	ww/dw	lipid/ TOC	ww/dw	lipid/ TOC	ww/dw	lipid/ TOC	ww/dw	lipid/ TOC	ww/dw	lipid/ TOC	ww/dw	lipid/ TOC
Maximum	53,765	1,143,936	0.04	0.07	0.10	0.06	0.10	0.11	0.39	0.11	0.05	0.16	0.21	0.17
Minimum[1]	23	1,062	82.10	76.04	2.68	3.06	232.24	115.22	5.08	2.06	115.48	167.70	3.04	4.35
North SWAC	12,729	733,646	0.15	0.11	0.18	0.08	0.42	0.17	0.62	0.14	0.21	0.24	0.35	0.49
South SWAC	2,492	112,624	0.76	0.72	0.37	0.23	2.14	1.09	1.07	0.30	1.07	1.58	0.61	0.28
Total SWAC	7,026	387,704	0.27	0.21	0.24	0.12	0.76	0.32	0.76	0.18	0.38	0.46	0.42	0.28
CH2M Hill Average[1]	10,648	484,000	0.18	0.17	0.20	0.10	0.50	0.25	0.66	0.16	0.25	0.37	0.37	0.25

Note: Measured BSAFs are the ratio of mean tissue concentration and sediment SWAC.
Regression model BSAFs are the ratio of tissue concentration predicted by logistic regression models from CH2M Hill (2010a) and sediment SWAC
CH2M Hill average value from CH2M Hill (2008).

[1] TOC not available. TOC-normalized values calculated using measured average TOC in the Lauritzen Channel (2.2%).

Table 3. Fish Tissue Risk-Based Calculation Parameters

Exposure Parameter	Units	CH2M Hill (2010b)	Alternative
THQ	unitless	1	1
BW	kg	70	70
AT	days	10,950	10,950
DDT RfD	mg/kg-day	0.0005	0.0005
EF	days/year	350	350
ED	years	30	30
$Frac_s$	unitless	0.5	0.1
IR_{fish}	g/day	85.1	42.5
CF	kg/g	0.001	0.001
Fish Tissue Risk-Based Concentration	mg/kg	0.86	8.59

Table 4. Results of DDT fish tissue and sediment RBC corrections

	Tissue RBC (µg/kg wet wt)		Sediment RBC (µg/kg dry wt)		
				Corrected RBC	
Receptor	2010 RBC[1,2]	Correction	2010 RBC[1,2]	(ww/dw) Model	(lipid/TOC) Model[3]
Ecological Risk-Driver					
Shiner surfperch	600	4,620	400	5,650	27,900
Forster's tern[4]	593	593	440	4,400	22,000
Double-crested cormorant[5]	882	882	700	2,345	5,854
Human Health Risk-Driver[6]	860	8,590	450		
All fish				123,000	955,000
Benthic fish				35,200	199,000

[1] Ecological values from CH2M Hill (2010a). Cormorant values are for females. Sediment RBCs are mean values.

[2] Human health values from CH2M Hill (2010b). Based on non-cancer risk HI = 1. Sediment RBC calculated using surfperch accumulation model.

[3] Corrected sediment RBC from lipid/TOC normalized model converted to dry wt basis using average TOC of 2.2%.

[4] Corrected sediment RBC for tern estimated using anchovy bioaccumulation logistic regressions.

[5] Corrected sediment RBC for cormorant estimated using all fish bioaccumulation logistic regressions.

[6] Corrected sediment RBC for human health calculated using both all fish and benthic fish bioaccumulation logistic regressions.

<u>Attachment B</u>

27201 Puerta Real, Suite 350
Mission Viejo, California 92691
Phone 949.347.2780
www.anchorqea.com

DRAFT MEMORANDUM

To:	Joe Kelly, President, Montrose Chemical Corporation	**Date:**	May 22, 2015
From:	Michael Whelan, P.E. and John Verduin, P.E. Anchor QEA, LLC	**Project:**	150754-01.01
Cc:	Kelly Richardson, Jeff Carlin and Steven Lesan, Latham & Watkins David Templeton, Anchor QEA, LLC		
Re:	Engineering Review of Sediment Remediation Assumptions and Costs Presented in Draft Focused Feasibility Study Former United Heckathorn Marine Sediment Site, Richmond, California		

INTRODUCTION

This technical memorandum presents a review of the Draft Focused Feasibility Study (Draft FFS) recently issued by the U.S. Environmental Protection Agency (USEPA) for cleanup measures at the Former United Heckathorn site in Richmond, California (Site; Figure 1). The Draft FFS presents the development and analysis of remedial alternatives for marine sediments that continue to be impacted by various contaminants of concern in the Lauritzen Channel (referred to herein as "the Channel"), following an initial cleanup attempt in 1996 and 1997.

This review focuses on engineering, design, and implementability issues relative to the USEPA's alternatives analysis, provides commentary on their screening of remedial alternatives, and points out areas where their assumptions and cost predictions appear to be incomplete or unrealistic. As such, we recommend that USEPA more completely screen the options that they have proposed, and consider additional options, as part of their finalization of the Draft FFS. As part of this review, we have developed conceptual-level opinions of probable cost for key aspects of the cleanup work. Costs, where provided, are intended as a Rough Order-of-Magnitude (ROM) level, as appropriate for the early and conceptual nature of the cleanup alternatives described.

EXECUTIVE SUMMARY

USEPA's Draft FFS for the United Heckathorn Sediment Site evaluates three alternatives for conducting further remedial activities in the Lauritzen Channel (in addition to a No Action scenario). All three alternatives are heavily weighted toward dredging the Channel; in the majority of the Channel (the East and West Side areas), USEPA only evaluated dredging. Two of the alternatives also included varying amounts of engineered capping applied to parts of the Northern Head.

Based on our experience with similar projects, the Draft FFS does not present a realistic analysis of the difficulties, complications, durations, and costs of dredging the Lauritzen Channel. Specifically, the Draft FFS:

- Envisions that most of the channel (90%) can be dredged in an open and unconstrained manner, although most of the channel poses hindrances that will slow down the dredging process and take significantly longer than stated in the Draft FFS.
- Underestimates the expected volume of sediment that would need to be removed from the Channel, based on an unrealistic description of how cleanup dredging is designed and implemented.
- Underestimates the costs of transport and disposal at an off-site and out-of-state location, as well as underestimating several other associated costs of the project.
- Largely overlooks the considerable degree of impacts to the public, environment, and community that would accompany a lengthy period of dredging and sediment transport.

As a result, we expect that the actual cost of designing and implementing a remedial dredging project in the Channel will be nearly twice the cost estimated in the Draft FFS, and that the work will take several months longer – potentially extending into a second construction season, given the annual regulatory dredging closure period for salmonids protection. Given these considerations, it seems imprudent for USEPA to emphasize dredging sediment quantities of this magnitude without a more comprehensive evaluation of alternative remedial approaches in the Channel. Alternative remedies, potentially combined with focused dredging of locally elevated chemical concentrations, is a reasonable and implementable course of action that bears further evaluation.

By only pursuing alternatives that are heavily weighted toward dredging, USEPA failed to properly assess the feasibility of other remedial approaches that are more cost-effective and which could significantly reduce environmental and community impacts associated with dredging. Engineered capping offers a much more cost-effective potential solution for contaminated sediments, by confining them permanently in place. Further, USEPA has noted the effectiveness and implementability of in-situ treatment of the sediment using activated carbon, as well as its even greater potential cost savings. Despite these benefits, engineered capping and in-situ treatment by activated carbon placement were only evaluated by USEPA for the Northern Head area and as a source control measure. In our opinion, both remedial approaches have potential to be used more widely in the Channel. Finally, on-site confined sediment disposal could be an attractive option for the Channel, because it lessens or eliminates the need for costly off-site hauling of sediment while providing usable uplands area. While USEPA briefly notes some limitations with these alternatives, those limitations have been successfully overcome at other sites nationwide which faced similar challenges, including project examples for which Anchor QEA has been involved with planning, designing, monitoring, and overseeing implementation. (See further detail on Anchor QEA's unique qualifications, below.)

In summary, it is imperative that the Draft FFS fully vet the alternatives of capping, in-situ treatment with activated carbon, and confined disposal, to inform the public and decision makers of all potentially feasible options, because of these alternatives' potential for effective remediation and cost savings, and because the three proposed dredging alternatives have numerous challenges of their own. To that end, we have provided rough-order-of-magnitude cost comparisons for key cost elements of engineered capping, in-situ treatment with activated carbon, and a conceptual confined disposal alternative for the site.

QUALIFICATIONS AND EXPERIENCE OF ANCHOR QEA

Anchor QEA provides this review and commentary as a national leader in designing and performing construction of a wide range of sediment remediation projects at sites similar to the Lauritzen Channel. Our review has been developed based on our experience with numerous successfully completed sediment projects in California, the West Coast, and nationwide, making our views an important addition to the project documentation and decision-making process. Relevant project experiences include the following example projects, for which we have provided design and construction management services:

- **San Diego Shipyards**. This project, currently ongoing with Anchor QEA acting as construction manager, involved dredging in two neighboring heavily used shipyards, with upland disposal of sediments and placement of sand layer in underpier areas.
- **Campbell Shipyards, San Diego**. This project involved localized dredging with upland disposal and engineered capping of undredged material. Work was sequenced so as to avoid impacting the active use of an adjoining Port terminal.
- **Rhine Channel, Newport Beach.** This project involved contaminated sediment dredging from a channel heavily used by private vessels, with barge transport of sediment and placement in a nearshore confined disposal site.
- **IR Site 7 and Middle Harbor Redevelopment, Port of Long Beach**. This project involved contaminated sediment dredging from a Port waterfront area, with sediment placement in a nearby nearshore confined disposal site.
- **Port Hueneme CAD, Oxnard Harbor District**. This project involved excavation of a submerged sediment disposal cell, use of excavated sand for beach nourishment, dredging of contaminated sediments from actively used Port and Navy wharves, and placement of sediment into the cell for permanent confinement.
- **Los Angeles River Estuary, Long Beach**. This project involved dredging of an industrialized river mouth with placement of sediments in a designated offshore area where they were covered with clean material.
- **East Waterway Deepening Project, Port of Seattle, Washington**. This project took place in a heavily used Port industrial waterway, and involved dredging and upland disposal of contaminated sediment. Operational constraints included dredging around vessel traffic and ongoing Port operations.
- **Terminal 4 Deepening, Port of Portland, Oregon**. Similar to the East Waterway project in Seattle, this project took place in a heavily used industrial area, with dredging and upland disposal of contaminated sediment. The project involved dredging around vessel traffic and ongoing commercial operations.
- **Confined Disposal Facilities for contaminated sediment at the Sitcum Waterway, St. Paul Waterway, and Hylebos Waterway at the Port of Tacoma, Washington**. Each of these projects involved active port terminal complexes, and dredging and sediment placement needed to be sequenced around ongoing industrial operations.

- **Esquimalt Harbor, Canada**. Project involves dredging, placement of residuals cover, and environmental monitoring - all accomplished in a heavily used harbor area.
- Similar efforts on nationally significant sediment remediation projects at the **Hudson River** and **Onondaga Lake** in New York State and **Fox River** in Wisconsin, among many others.

The potential options for sediment remediation discussed in the Draft FFS and in this memorandum are all activities which Anchor QEA has successfully designed and overseen construction for other projects. Our experience and perspective allows for a realistic opinion of the cost factors applicable to these alternatives conducted as a USEPA cleanup program, specifically in the Bay Area.

PROJECT BACKGROUND

Marine sediments impacted by dieldrin and DDT (among other contaminants of concern) have been present historically in the Lauritzen Channel and Parr Canal adjoining the former United Heckathorn site and were addressed under a 1994 Record of Decision (ROD) and subsequent Consent Decrees (CD), from USEPA (1994). Upland soils were addressed as separate remedial actions. Figure 2 depicts the Lauritzen Channel along with its recently surveyed bathymetry.

In 1996 and 1997, the Montrose Chemical Corporation of California, Inc. (Montrose Chemical), performed remedial actions in the Lauritzen Channel and Parr Canal, pursuant to a USEPA CD, as follows:

- Mechanical dredging of 107,000 cubic yards (cy) of sediments (in-situ volume) was conducted, primarily from the Lauritzen Channel (with some from the adjacent Parr Canal), with on-site dewatering, off-site transport by rail, and landfill disposal in Utah.
- Dredging was designed to remove younger bay muds from the Lauritzen Channel and Parr Canal, down to the underlying, older bay mud. The site remediation goal for sediments was 590 parts per billion (ppb) for DDT.
- After reaching the design depth, a 6- to 18-inch layer of clean sand was placed over dredged areas in the Channel and in underpier areas.

- Construction of a cap over upland portions of the facility was completed, consisting of reinforced concrete in some areas and geotextile fabric and gravel in others.

Following the completion of remedial actions in 1997, post-cleanup surface sediment concentrations were measured, and a period of post-remediation monitoring began, with a frequency of 5 years (as needed) between monitoring events. Post-remedial monitoring results are documented in a series of 5-year review reports, prepared by USEPA (2001, 2006, and 2011). In response to the findings in these reports, USEPA has performed further site studies to evaluate possible further cleanup options. For example, a Source Identification Study Report, prepared by CH2M Hill (2014) on behalf of USEPA evaluated available monitoring data and possible sources of recontamination.

To address the continued presence of dieldrin and DDT in the Lauritzen Channel, USEPA issued a Draft FFS for the Site, which describes four cleanup alternatives:

- Alternative 1: No action
- Alternative 2: Dredging of the East Side and West Side areas; capping the Northern Head, under piers, and side slope areas; and source control measures
- Alternative 3: Dredging of the East Side, West Side, and portion of the Northern Head; capping the remainder of the Northern Head, under piers, and side slope areas; and source control measures
- Alternative 4: Dredging of the East Side, West Side areas, and Northern Head; capping under piers, and side slope areas; and source control measures

Aside from the No Action alternative, the list of alternatives in the Draft FFS is focused almost entirely on the concept of dredging impacted sediments, with off-site transportation to an out-of-state landfill. The application of engineered capping is confined only to the Northern Head. On-site confined disposal of sediments was eliminated as an option for this Site. In-situ treatment was also eliminated as an option for the Site.

In the next section of this memorandum, we explain how certain key assumptions and expectations described for dredging in the Draft FFS, are unrealistic, and significantly underestimate the time, difficulty, and cost of sediment dredging with off-site disposal. Later in this memorandum, we discuss why the alternative measures of engineered capping, in-situ treatment using activated carbon, and on-site confined sediment disposal are worthy of further review by USEPA.

DREDGING ASSUMPTIONS UNDER-PREDICT COMPLICATIONS AND COSTS

All of the active cleanup options considered in the Draft FFS are largely centered on removal of contaminated sediments by dredging, with disposal at an off-site, out-of-state, permitted landfill. The total volume of sediment removal, and the overall rate, duration, and costs for the work, were estimated for the various options. For Alternative 4, in which virtually all of the Lauritzen Channel is dredged (except for underpier areas and side slopes), the total predicted dredged volume presented in the Draft FFS was 66,000 cy, and the total estimated cost was $22,711,303. This amounts to a price of approximately $344 per cubic yard. (Alternatives 2 and 3 had lesser dredging volumes and proportionately lower costs.) This total price per cubic yard—intended to be inclusive of all project elements, including permitting, design, implementation, and monitoring—appears to be low compared to recently completed projects in the United States and California, which typically end up with prices approximating $450 or more per cubic yard (such as the recently completed South Shipyard Sediment cleanup in San Diego, which had a total cost between $420 and $440 per cubic yard).

In our estimation, and given our experience with dredging projects similar to those proposed in the Draft FFS, the actual design and implementation of a remedial dredging project has been considerably oversimplified in the analysis presented in the Draft FFS. As a result, we expect that redredging the Lauritzen Channel would be significantly more time-consuming and expensive than the Draft FFS envisions. The following sections present a closer look at the dredging design process, and the actual construction costs that should be expected, focusing in particular on three critical areas of USEPA's analysis:

- The difficulty of dredging in the Lauritzen Channel has been underestimated; as a result, USEPA's dredging rates and costs are overly optimistic.
- The expected volume of sediment that would be removed from the Lauritzen Channel has been underestimated in USEPA's analysis. Certain practical aspects of the dredging design and construction process will inevitably lead to a greater mass of sediment being removed.

- Actual sediment disposal costs may be considerably higher than those assumed in the USEPA's analysis.

Difficulty of Dredging in Lauritzen Channel Has Been Underestimated

Remedial construction would have to work around shoreline structures and berthed vessels, and would need to be scheduled around ongoing vessel traffic and facility activities in the Lauritzen Channel to avoid potential costly impacts on industrial and commercial operations. This will have a sizable impact on the dredging process, to a degree that is severely underestimated in the Draft FFS.

The Draft FFS Underestimates the Extent of Constrained Dredging

Because the Lauritzen Channel is only 200 to 250 feet wide, there is little room for vessels to maneuver. As a result, we anticipate that dredging within the channel will encounter numerous and frequent delays and disruptions. USEPA has separated the dredging area into two categories: "open area" dredging, and "tight area" dredging, and has estimated the rate and cost of dredging for each type of area.

The Draft FFS makes the unsupported assumption that 10% of the dredging volume in the Lauritzen Channel would qualify as "tight area" dredging, and the remaining 90% qualifies as "open area" dredging. This assumption is intended to recognize the complicating effect of adjacent structures, but in our experience the 90%/10% split greatly under-represents the extent of impacts that would be posed by marine structures and active vessel operations at the berths and within the relatively narrow channel itself. This is especially true given the considerable marine activities that currently take place in the Lauritzen Channel, as summarized in the Sediment Transport Study (CH2M Hill 2013):

> *The present description of vessel activity is based upon conversations with vessel and terminal operators in the area and anecdotal observations. The most common large bulk carrier vessels into the Lauritzen Channel are of the Handysize design between 40,000 and 55,000 Deadweight Tons (dwt) going to the Levin facility. The typical vessel docks and departs with two tugs. The tugs are characterized as tractor tugs.* […]

> *Manson Construction Company has its main San Francisco Bay berthing and staging facility on the west side of the Lauritzen Channel. Manson generally has on the order of 6 to 10 unpowered crane and construction barges anchored with spuds in the channel. These barges are moved with tugs in the 1000 hp class. The values presented herein will be further investigated.*

As is shown on Figure 3, constraints on the dredging process will result from a number of factors, including:

- Proximity of side slopes, wharves, and structures;
- Positioning of moored vessels and barges, which typically cannot be changed and can cause delays to the dredging process; and
- Allowance for marine traffic to move through the area of dredging, which requires movement of dredging barges, support vessels, and in-water environmental controls (turbidity curtains).

Thus, the amount of dredging that qualifies as constrained is much greater than 10%, and the proportions may very well be reversed, as there is very little of the channel that would qualify as open. For this estimation, it is realistic to assume that 75% of dredging is constrained, and that only 25% (and possibly less) is unconstrained, or open.

The Draft FFS Overestimates Dredging Production Rates

Based on our experience with remedial dredging, the Draft FFS has overestimated the rate that can be expected for dredging in the Lauritzen Channel. Although the assumptions of a 4-cy bucket and continuous 24-hour 7-day working schedule are reasonable, the production will be slowed by additional variables, such as dredge cycle time and the percentage of uptime (the percentage of in-water time that the dredging equipment is actively dredging, which is a function of pauses for movement, shift changes, water management, equipment maintenance, regular repairs, etc.) that USEPA has failed to account for in the Draft FFS. With these expectations, Table 1 presents updated estimates of production rates for dredging.

Table 1
Estimation of Dredging Production Rates

Category	Open-Water Dredging Areas	Constrained Dredging Areas
Number of Dredges Operating	1	1
Dredging Bucket Size	4 cubic yards	4 cubic yards
Dredging Schedule	Continuous (24/7)	Continuous (24/7)
Bucket Cycle Time	2.5 minutes	4 minutes
Dredging Uptime	60% of time	50% of time
Bucket Recovery	70% of bucket volume	70% of bucket volume
Estimated Daily Dredge Production Rate	**970 cubic yards/day**	**500 cubic yards/day**

For comparison, the Draft FFS indicates dredging production rates of 1,500 cy per day for open areas, and 1,250 cy per day for constrained (tight) dredging areas.

Dredging Volumes Have Been Underestimated

USEPA estimated the volume of surficial sediments (Young Bay Mud [YBM]) in the Lauritzen Channel to be approximately 66,000 cy, based on the thickness of Young Bay Mud sediments observed in a series of sediment cores obtained in 2013. Their evaluation assumed that the Young Bay Mud is the material that is impacted by dieldrin and DDT, and thus is the volume to be targeted for dredging. What the Draft FFS appears to have overlooked, however, are some key practical aspects of the dredging design and construction process which, in implementing the identified dredging alternatives, will inevitably lead to a greater mass of sediment being removed than the 66,000 cy of YBM.

To remove the targeted YBM material, an implementable dredge plan needs to identify discrete target dredging depths, selected to completely encompass the targeted sediments. Because an irregular mass of targeted sediments needs to be converted into a series of flat, bounded dredging areas, the overall volume of material removal would increase. As an example of how dredging would need to be designed, we have developed a conceptual dredge plan for the Lauritzen Channel, shown on Figure 4. This dredge plan also includes the removal of materials from adjoining side slopes.

The conceptual dredge plan shown on Figure 4 was developed by identifying the required depth of sediment removal at each of the 2013 sediment cores (based on the presence of YBM and extent of cleanup criteria exceedances for dieldrin and DDT), and selecting a representative target depth for dredging in various areas of the channel. Due to localized irregularities in the sediment thickness and target depth, and the need to divide the project area into manageable subunits, it is necessary to establish target dredge depths that are frequently deeper than the YBM depths indicated at individual core locations. Dredging depths are frequently determined by taking the necessary depth of sediment removal, and rounding up to the nearest foot deeper.

The contractor will also remove an additional quantity of overdredge volume from below the target dredge depths to ensure that they have fully removed the targeted material. An overdredging allowance needs to be anticipated to ensure that the neatline volume is fully removed, accounting for the accuracy of the dredging process. For remedial dredging projects, specified overdredging allowances are typically in the range of 1 to 2 feet.

The conceptual dredge plan shown on Figure 4 results in the following approximate dredging volumes for the full extent of the Lauritzen Channel:

- 70,000 cubic yards for dredging to the targeted elevations and side slopes
- Plus 10,000 cubic yards representing 1 foot of overdredging
- Equaling approximately 80,000 cy total dredging volume

After dredging to design grades is completed in a portion of the site, subsequent sampling will be needed to confirm whether remedial goals have been accomplished by the dredging. The dredging process will result in some amount of residual impacted sediment which will require management, per U.S. Army Corps of Engineers (USACE) guidance (USACE/ERDC 2008). The residuals could result from settling of sediment that was temporarily suspended by the dredging process, from the presence of chemically impacted sediments to depths greater than anticipated by the dredging plan, or a combination of both factors.

Any remaining elevated concentrations in post-dredge conformational samples will require that a decision be made as to how to best manage the residuals. In some cases, when chemical exceedances are marginal, or the residual layer is relatively thin, placing a clean sand cover over the dredged surface can be an acceptable way of re-establishing cleanup goals for the sediment surface. The Draft FFS envisions laying down a 6-inch sand layer after dredging is completed. However, in cases where a greater thickness of material remains in place, or when chemical exceedances are more definitive, it may be more appropriate to perform an additional dredging pass in the region represented by the sample(s). These would add further to the overall volume being dredged. For example, if an additional dredge cut of nominal 3-foot thickness were made over one-half of the dredging area, that would equate to another 17,000 cy. While this amount of residuals dredging may not be necessary, it is important to leave some allowance in estimates for a potential second pass. Here we have applied an additional 5,000 cy to the volume estimate to represent potential residuals dredging.

The total amount of sediment predicted to be produced by dredging the entire Lauritzen Channel—equivalent to Alternative 4 in the Draft FFS—is therefore 80,000 cy (first pass of dredging) plus 5,000 cubic yards (residuals management) for a total of **85,000 cy.**

Dredging in the Lauritzen Channel in 1996-1997 encountered a large amount of debris that needed to be handled separately from the sediment. This is not unusual for an industrialized waterway, and is likely to be a factor if further dredging is completed. It is not clear from the Draft FFS how the potential of debris is specifically factored into the dredging cost estimates, although the amount of debris was estimated as being 0.1% of dredging volume for limited access areas and 1% of dredging volume for open water areas. In our experience, these expectations are far too low. Typical project experience in a heavily industrialized and frequently used channel such as the Lauritzen, and specifically our recent experiences with projects in San Diego and the Northeast, indicate that debris totals would be closer to 2% of dredging volume for open water areas and 10% of dredging volume for areas below piers.

Given these breakdowns of dredging conditions, and our estimation of dredging rates, Table 2 presents estimated durations for the dredging project:

Table 2
Summary of Estimated Duration of Dredging in Lauritzen Channel[1]

Quantity	Open Areas (25% of total)	Constrained Areas (75% of total)	Total
Dredge Volume	21,000 cy	64,000 cy	85,000 cubic yards
Estimated Dredging Rate	970 cy/day	500 cy/day	Combined rate
Estimated Dredging Duration	22 days	128 days	150 days (5 months)[2]

Notes:
1. Based on dredging of the entire Lauritzen Channel (as presented in Alternative 4 in the Draft FFS).
2. Redredging or additional dredging could extend the construction time, adding one or more months to overall project duration.
cy = cubic yards
n/a = not applicable

This duration is much longer than the 40-day duration estimated for Alternative 4 by the Draft FFS. (Similarly proportionate conclusions will apply to Alternatives 2 and 3, which involve marginally less dredging.) In fact, depending on potential slowdowns, stoppages for wharf operations, additional residuals dredging, or other variables, a project that dredges the entire Lauritzen Channel could extend into a second construction season. In the Bay Area, the regulatory environmental work window for dredging activity spans from June 1 to November 30, for the protection of salmonids. If in-water construction work threatens to extend beyond the regulatory environmental work window, consultation with the resource agencies (National Oceanic and Atmospheric Administration Fisheries, U.S. Fish and Wildlife Service, and California Department of Fish and Wildlife) would be required to determine if dredging can continue without adversely affecting listed species. If the resource agencies determine that work is not allowed to occur past the environmental work window, dredging would need to temporarily cease, and the resulting shut-down for biological protection would likely require a partial or full demobilization and second mobilization to the site once the environmental work window reopens. Alternatively, if dredging was allowed to continue past the environmental work window, the resource agencies may require biological monitoring to be conducted, which would increase project costs.

These issues illustrate another important factor that was given little consideration in the Draft FFS, the potential impact dredging and off-site transportation of sediment would have on the community. While the Draft FFS (Table ES-1) mentions potential community risks due to "increased levels of traffic, dust, noise, and odors", there is little to no discussion of the fact that dredging and off-site transportation increases many of those risks by orders of magnitude compared to remedial solutions that do not require hauling dredged sediment away by road or by rail. Transporting over 100,000 tons of sediment, plus an additional amount of debris, over a distance of hundreds of miles, clearly poses a number of impacts near site ingress/egress access points as well as along the entire length of the selected haul routes. The transportation process will also result in a sizeable increase to project emissions. It should also be noted that the noise and odor impacts arising from dredging activities at the Site will be worsened by the longer duration necessary to complete the work, as noted previously.

Longer construction duration directly impacts the unit costs for dredging, which are determined based on the length of time that equipment and personnel need to be on-site conducting the work. The corresponding predicted increase in unit costs is reflected in Table 3, presented at the end of this section. The longer duration also would increase impacts to the community originating from the dredging project, both at the site where the dredging equipment would be working, in the surrounding areas through with trucks or rail lines would pass, and in the surrounding area of dewatering facilities.

Sediment Disposal Will Be More Costly Than Envisioned by Draft FFS

Based on the total DDT concentrations in the Lauritzen Channel, the dredged material would be considered hazardous waste in California (California Code of Regulations, Title 22, 66700) and would need to be disposed of at a permitted hazardous waste landfill facility. The Draft FFS anticipates out-of-state sediment disposal, which is consistent with the fact that the sediment removed in 1996/1997 was hauled to a facility in Utah. However, an optimistic unit cost of $99.90 per ton was assumed for transportation and disposal in the Draft FFS. Based on a preliminary investigation of potential receiving sites and transportation costs, it is anticipated that the actual costs may be higher and are highly dependent on actual production rates achieved during construction.

A unit cost of less than $99.90 per ton assumes a steady rate of production and transport by rail. However, a number of variables exist that will make this best case scenario difficult to achieve. The Lauritzen Channel is an active marine area with daily commercial/industrial activity. Rates of production will be based on the selected contractor's means and methods for dredging, de-watering, transport, and disposal. The logistics associated with efficient use of rail transport requires adherence to a set and regular production schedule; any disruptions to production will lessen the effectiveness of this transportation option, and it may prove necessary to mobilize two dredges in order to maintain the rates, increasing overall costs. Conversely, rail issues can backlog a dredging project. It is not uncommon for haul cars to be delayed due to rail traffic or availability. Therefore, it is not necessarily realistic to assume that rail transportation will apply to the project. Transport by truck is much more flexible and results in a slight increase in unit costs.

Based on our discussions with local and regional waste disposal representatives (specifically, Clean Harbors and Republic Services), we recommend assuming a unit cost of $110 per ton for estimating costs of transport by truck to a permitted in-state location. Note, however, that prices could vary to as high as $125 per ton based on specific operational considerations and disposal locations. We understand from regional waste disposal representatives that for out-of-state disposal, the difference between rail and truck transport has much more impact on the project cost than would be the case for in-state disposal. When hauling sediments out of state, transport by rail could be estimated at $110 per ton, while out-of-state transport by truck could vary to as high as $250 per ton.

Hazardous waste material can be classified as Resource Conservation and Recovery Act (RCRA) or non-RCRA. In the 1994 ROD for the United Heckathorn Site, USEPA determined that contaminated marine sediments from the site would not be regulated under RCRA. This is likely based on the fact that USEPA only considers dieldrin and DDT-based wastes to be RCRA waste if they are discarded unused (i.e., spilled) or in their pure form (i.e., 100% of that chemical). Neither condition appears to apply at this site, but if the material were to be re-classified as RCRA waste, then treatment to reduce concentrations would likely be necessary, which would likely require incineration, and result in disposal costs that may be as high as $650 per ton.

In addition to undervaluing the dredging volumes and operational costs, other elements of the work also appear to be underestimated in terms of costs and community impacts, including the following:

- Work planning, project management, and design
- Project mobilization and demobilization
- Installation and deployment of turbidity curtains
- Water quality monitoring
- Bathymetric surveying
- Treatment of water generated by the dredging process
- Removal of residual sediments from municipal storm drain

Table 3 presents a compilation of Anchor QEA's adjusted estimated costs associated with dredging and construction at the site, as compared to the cost assumptions presented in the Draft FFS. It can be seen that the total of Anchor QEA's estimated costs is nearly double what is presented in the Draft FFS, resulting both from the increased dredging volume, the increased project duration, and other cost factors that appear to have been under-represented in the Draft FFS. The costs in Table 3 pertain to Alternative 4, in which the entire Lauritzen Channel undergoes dredging. A similar comparison of costs would also apply to Alternatives 2 and 3.

Table 3
Comparison of Construction Costs for Sediment Dredging in Lauritzen Channel

Activity	Item/Activity Number	Costs Presented in Draft FFS				Revised Costs Recommended by Anchor QEA				Comments
		Quantity	Unit	Unit Cost	Cost	Quantity	Unit	Unit Cost	Cost	
Mobilization	3.2	1	Lump Sum	$265,000	$265,000	1	Lump Sum	$900,000	$900,000	Combined total of mobilization plus demobilization should be close to 5% of construction cost. May need second mobilization after environmental work window reopens.
Turbidity Curtains	3.4	1	Lump Sum	$72,900	$72,900	1	Lump Sum	$250,000	$250,000	Draft FFS assumptions are not entirely clear. Recommended number is more consistent with likely size and length of curtains needed at this site, and is based in part on current work taking place at San Diego Shipyards site.
Water Quality Monitoring	3.5	1	Lump Sum	$170,910	$170,910	1	Lump Sum	$750,000	$750,000	Longer construction duration (5 months) than assumed by Draft FFS.
Mechanical Dredging: Constrained Areas (Tight Areas)	3.6	6,600	Cubic Yard	$22.86	$150,876	64,000	Cubic Yard	$67	$4,290,000	Increased volume and significantly slower dredging production rate estimated
Mechanical Dredging: Open Areas	3.6	59,400	Cubic Yard	$16.73	$993,762	21,000	Cubic Yard	$26	$550,000	Less of the dredging occurs in open water than what the Draft FFS assumed
Reagent Mixing and Stabilization of Sediments	3.6	66,000	Cubic Yard	$18.45	$1,217,700	85,000	Cubic Yard	$18.45	$1,570,000	Activity item replicated from FFS unit, with cost unchanged. Applicable quantity has been increased.
Loading and Transport of Sediments (to Handling Area)	3.6	108,499	Ton	$10.67	$1,157,684	140,250	Ton	$10.67	$1,500,000	Activity item replicated from FFS unit, with cost unchanged. Applicable quantity has been increased.
Off-loading and Placing Dredge Material on Mixing Pad	3.6	66,000	Cubic Yard	$5.00	$330,000	85,000	Cubic Yard	$5.00	$425,000	Activity item replicated from FFS unit, with cost unchanged. Applicable quantity has been increased.
Transport and Off-Site Disposal of Sediment	3.6	108,449	Ton	$99.90	$10,834,055	140,250	Ton	$110	$15,430,000	Haul by rail to out-of-state facility (Utah).
Bathymetric Surveys	3.6	1	Lump Sum	$36,000	$36,000	1	Lump Sum	$84,000	$84,000	Costs of surveys conducted on recent and ongoing remediation project in San Diego were approximately $7,000. The Draft FFS assumes 12 surveys will be taken.
Debris Removal	3.6	978	Ton	(unclear)	(unclear)	12,900	Ton	$150	$1,940,000	Assumes 2% of open water dredge volume and 10% of constrained dredge volume. 1.9 T/cy unit weight.

Activity	Item/Activity Number	Costs Presented in Draft FFS				Revised Costs Recommended by Anchor QEA				Comments
		Quantity	Unit	Unit Cost	Cost	Quantity	Unit	Unit Cost	Cost	
Removal of Residual Sediment from Municipal Storm Drain System	3.6	1	Lump Sum	$299,900	$299,900	1	Lump Sum	$600,000	$600,000	Cost assumed by the draft FFS are unclear. Prior estimates provided to the City of Richmond by USEPA for removal of storm drain sediments were in the range of $600,000, as mentioned in USEPA's September 2013 status update (USEPA, 2013).
Dredging: Water Treatment	3.7	3,740,132	Gallons	$0.07	$261,809	4,000,000	gallons	$0.10	$400,000	--
Demobilization	3.9	1	Lump Sum	$150,000	$150,000	1	Lump Sum	$700,000	$700,000	Combined cost of mobilization plus demobilization should be close to 5% of construction cost. Also may need an initial partial site demobilization if construction needs to be suspended for environmental work window.
		TOTAL OF ROM COSTS ABOVE			**$16,840,600**	**TOTAL OF ROM COSTS ABOVE**			**$29,389,000**	
Net Increase in Costs, Alternative 4, resulting from cost elements listed above									$13,448,404	
USEPA Total Estimate of Costs for Mobilization, Construction, Operation, Maintenance, and Demobilization									$18,873,425	
Adjusted Total Estimate of Costs for Mobilization, Construction, Operation, Maintenance, and Demobilization									**$32,321,829**	
Construction Project Add-Ons										
Performance and Payment Bonds (2%)									$646,437	
Technical Design (6%)									$1,939,310	Per USEPA (2000) cost estimating guidance
Project Management and Overhead (5%)									$1,616,091	Per USEPA (2000) cost estimating guidance
Construction Management (10%)									$3,282,183	Assumed 10% due to complexities of site and ongoing site operations.
TOTAL PROJECTED ROM COST FOR CLEANUP ALTERNATIVE 4 (DREDGING OF CHANNEL)									**$39,755,850**	
USEPA Total ROM Cost Estimate, Alternative 4 (for comparison)									$22,711,303	
Resulting unit cost per cubic yard dredged									**$468**	

Notes:
cy = cubic yards
FFS = Focused Feasibility Study
ROM = Rough Order-of-Magnitude

Because the Draft FFS so significantly underestimates the actual cost, community impacts, and duration of dredging, it is important that USEPA more fully evaluate other remedial approaches in the Lauritzen Channel, even though they were screened out in the Draft FFS process. The next four sections explore remedial strategies that warrant further evaluation in the Draft FFS: the further application of engineered capping, application of granular activated carbon (GAC) as in-situ treatment, hybrid alternatives that combine dredging and in-situ treatment, and on-site retention of sediments.

EXPANDED USE OF IN-SITU CAPPING WARRANTS FURTHER EVALUATION

The only portion of the Lauritzen Channel considered by USEPA for engineered capping is the Northern Head. Capping was not evaluated in the remainder of the Channel, apparently because of perceived incompatibility with vessel activity and industrial uses. However, given the high dredging and disposal costs applicable to the alternatives presented, the extended time period over which dredging activity would need to occur, and the resulting community impacts arising from dredging and off-site transportation of sediment, a closer look at in-situ capping in the West and East sides of the Channel is warranted.

As the Draft FFS notes, the Lauritzen Channel sees a variety of ongoing industrial uses and vessel traffic. The East side of the Lauritzen Channel is used by the Levin Pier and LRTC facility (as shown on Figure 2), where relatively deeper water exists; contours near the Levin Pier reach depths of -35 to -36 feet mean lower low water (MLLW). A significant portion of the Lauritzen Channel's chemically impacted sediments are present in this area of deep water, to thicknesses of approximately 5 to 6 feet, meaning that this area would likely contribute approximately 40,000 cubic yards to the estimated dredge volume. Similar considerations appear to apply to the West side of the Channel, where shallower water depths are present, ranging from approximately -10 feet to -25 feet MLLW and sloping gradually deeper away from the shoreline (Figure 2). This area is currently utilized by Manson Construction for equipment berthing and storage.

In the interests of fully analyzing remedial options at this site, and recognizing the severe costs and community impacts posted by dredging and off-site sediment disposal, we believe it is appropriate to consider an expanded use of in-situ capping—not only for the Northern

Head, but also for the West and East sides of the Channel. The following sections explore the engineered cap option in greater detail.

Components of Engineered Cap within Lauritzen Channel

The required thickness of an engineered cap depends on a variety of factors, such as the rate of transmission of water upward through the sediments, and the degree to which a surficial layer of protective armoring is needed.

For our estimating purposes, it is assumed that 12 to 18 inches of clean sand and gravel material would suffice to confine underlying contaminants and provide a filter layer for the armor layer discussed below. Further design-level analysis will likely conclude that less material would be sufficient, and the inclusion of absorptive components, such as activated carbon, to enhance the chemical protectiveness of the cap, may reduce the necessary thickness further. As an example, in the Northern Head, the Draft FFS envisions a three-inch activated carbon layer and six-inch sand layer, for a total thickness of nine inches of cap layer.

A permanent engineered cap would need to utilize armoring to protect it against propeller wash-induced erosion from passing vessels. The Draft FFS envisions a 12-inch-thick surficial armor layer in the Northern Head, where vessel traffic is expected to be relatively light. In the West and East sides of the channel, and in particular the Levin Pier in the East side of the Channel, larger vessels and erosive forces, may apply, so in these areas we have conservatively estimated that larger armor stone would be needed, such as a 2- to 2.5-foot-thick armor layer consisting of 1 to 1.75-foot stones. This armor layer, if placed over a 12- to 18-inch layer of clean sand and gravel, would result in a total projected thickness of 4 feet for the engineered cap. It is possible that smaller stone sizes and thicknesses would be sufficient for cap protection; this would need to be determined through a design analysis.

The engineered cap would be intended for long-term functionality, and would need to be verified through a program of long-term cap monitoring, including regularly scheduled bathymetry surveys to ensure the cap is not eroding.

Projected Costs for Engineered Cap in West and East Sides of Channel

Table 4, below, presents a rough order-of-magnitude cost breakdown for the construction of an engineered cap in the West and East sides of the Channel. The Northern Head is not included in this cost table as the Draft FFS already envisions potentially capping this area.[1]

Table 4

Comparative Costs for Engineered Cap in West and East sides of Lauritzen Channel[1]

Item	Quantity	Unit	Unit Rate	Cost
Additional Equipment Mobilization	1	Lump Sum	$500,000	$500,000
Additional Design and Permitting	1	Lump Sum	$1,000,000	$1,000,000
Place Clean Sandy Gravel (1.5 feet)	22,000	Tons	$35	$770,000
Place Armor Stone (2.5 feet)[1]	42,000	Tons	$50	$2,100,000
Long-Term Monitoring and Surveys	5	Episodes	$150,000	$750,000
			Total	$5,100,000
			Contingency Factor (35%)	$1,800,000
			ESTIMATED ROUGH-ORDER-OF-MAGNITUDE COST	**$6,900,000[3]**
Costs saved, for comparison				
Dredging not necessary for sediment that is capped[1]	70,000	Cubic Yard	$468[2]	**$32,760,000[3]**

Notes:
1. Approximately 6 acres of area in West and East sides of Channel. Does not include the Northern Head, which is already being considered for capping under Alternative 2 in the FFS.
2. Unit price of $468 per yard is based on the costs presented earlier, for dredging, treatment, transport, and disposal; in Table 3.
3. Costs are Rough-Order-of-Magnitude and presented for feasibility-level, comparative purposes only. The project needs to undergo a full design process before numbers can be refined. Consultant makes no warranty, express or implied, that the cost of the work will not vary from these cost values.

For the purposes of comparison, capping the west and east sides of the channel would cover approximately 70,000 cubic yards of sediment that would otherwise need to be dredged. Using a unit price of $468 per cubic yard removed to represent the costs of the dredging, transportation, and sediment disposal process (as developed in Table 3), this would equate to approximately $32,760,000 saved.

[1] However, capping the Northern Head appears to be an appropriate and technically feasible option.

Consistency with Ongoing Vessel Usage in Channel

USEPA appears to have ruled out capping in the Channel's East and West sides due to perceived conflicts with vessel berthing and related industrial uses. However, even with water depths made four feet shallower by placement of an engineered cap, the resulting depths will still allow for berthing of vessels, barges, and equipment, and ongoing industrial activities, even if some of those activities need to be modified because of the shallower depths. It is unclear whether LRTC has any specifically permitted depth authorizations within the Channel and alongside the Levin Pier. Nevertheless, it is recognized that ongoing vessel activities may need to be accommodated at the Levin Pier,[2] and that shallower water depths could impact vessel operations, potentially even precluding certain types of berthing and marine activity at the Pier.

USE OF IN-SITU SEDIMENT TREATMENT USING ACTIVATED CARBON WARRANTS FURTHER EVALUATION

Another in-situ remediation alternative involves *in-situ* treatment and remediation of sediment by applying treatment amendments directly to the sediment surface to promote absorption and immobilization of the contaminants (such as dieldrin and DDT) that are dissolved in sediment porewater. GAC has successfully been used for this purpose on a number of sites in North America and in Europe, as summarized in a recent study by Patmont, *et al* (2014), which has been provided as Attachment A to this memo. The GAC offers the advantage of providing an absorptive component to the sediment; by absorbing dieldrin and DDT molecules, the biologically available amount of both compounds is

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[2] One approach would be to perform limited dredging near the Pier so as to provide a deeper bottom surface upon which the engineered cap can be constructed. For example, targeting a final bottom elevation no higher than -30 feet MLLW along the face of the Levin Pier could be appropriate because this is the water depth that is currently authorized by the U.S. Corps of Engineers in the adjoining Santa Fe Channel. Approximately 8,000 cy of sediment would need to be dredged from the areas depicted on Figure 5 near the Levin Pier, to a depth of -36 feet MLLW to accommodate construction of a four-foot-thick engineered cap, while keeping the bathymetry below -30 feet MLLW (with a two-foot buffer depth to account for cap over-placement tolerances). At a predicted unit price of $468 per cubic yard dredged (as developed in Table 3), and applying a 35% contingency factor, the dredging of 8,000 additional cubic yards would add approximately $5 million to the overall project cost—still well below the cost of dredging the entire channel, and still significantly reducing the amount of sediment that would be transported off site.

lessened. Furthermore, GAC provides excess capacity to absorb contaminants that may be deposited on the floor of the Channel from external or continuing sources.

The concept of treating sediment by adding GAC, is fundamentally different from the engineered cap concept, because it is expected that the GAC material will be redistributed within the Channel over time by the same forces and currents that redistribute the sediment itself. Because the GAC material is added to and becomes integral to the overall sediment mass, some of which is likely mobilized by events, forces, and activities in the Channel, the GAC does not need to be confined permanently in place in the manner of an engineered cap. For this reason, no protective armoring is necessary in order to treat the sediment in-situ with GAC.

The technique, and its use at sites with vessel activity or erosive forces, has been the topic of considerable study. Through sediment stability tests conducted in the laboratory, Zimmerman, et al. (2008) demonstrate that sediments mixed with activated carbon do not adversely impact the stability of surface sediments. For a San Francisco Bay site, hydrodynamic modeling was used to estimate the maximum bottom shear stress encountered at the site due to natural forces. Physical testing demonstrated that critical shear stress for incipient particle motion were not significantly impacted by the application of activated carbon (Zimmerman et al. 2008).

In a pilot study field test, GAC treatment was applied to a sediment plot within San Francisco Bay. Results demonstrated 34% less PCB uptake and 24% less PCB bioaccumulation when compared to untreated sediment. Seven months after treatment, the decreases in contaminant uptake increased to 62% in uptake and 53% in bioaccumulation, indicating a trend of long-term effectiveness of this alternative remedial solution (Cho et al. 2007).

Further, independent analysis of Channel sediments at the Site appears to confirm the efficacy of activated carbon as a remedial measure. Tomaszewski, et al (2007) determined that, "because of [the] Lauritzen Channel sediment characteristics, adding small amounts of highly sorptive activated carbon to the sediment likely would have a significant effect on the portioning and availability of DDT." The sampling and analysis of Lauritzen Channel sediments by Tomaszewski, et al. further confirmed the potential for application of activated carbon to manage any residual DDT contamination remaining from the prior remedial action. USEPA incorporated the findings from this report into the Draft FFS.

While the Draft FFS acknowledges the efficacy, implementability and relatively low cost of activated carbon, and its use as part of an enhanced "active" cap, it restricts further evaluation of active cap materials to the Northern head of the Lauritzen Channel in Alternative 2 and 3, and as a source control measure. Such limitations are not explained and appear unfounded given the success of activated carbon at other similarly situated Sites.

To achieve this form of in-situ treatment, GAC would be applied to the sediment surface in a thin layer, after which it will mix into the sediment and potentially undergo localized redistribution along with the sediments, as described above.. Various procedures and products have been developed to facilitate the placement process such that GAC can be administered to the sediment without floating into the water column. Most commonly, these include proprietary products such as SediMite™ and AquaGate™, which are specifically designed to sink in the water column while also providing additional resistance to being resuspended by erosive forces. Figure 6 and Table 5 presents ROM-level costs for the application of a typical GAC-related product, based on project experience and case histories summarized in Patmont, et al (2014).

Table 5

Comparative Costs for Application of GAC throughout Lauritzen Channel [1]

Item	Quantity	Unit	Unit Rate	Cost
Additional Equipment Mobilization	1	Lump Sum	$500,000	$500,000
Additional Design and Permitting	1	Lump Sum	$500,000	$500,000
Granular Activated Carbon product[2]	7	Acre	$75,000[3]	$525,000
Place GAC throughout channel	7	Acre	$100,000[3]	$700,000
Long-Term Monitoring and Surveys	5	Episodes	$150,000	$750,000
			Total	$3,000,000
			Contingency Factor (35%)	$1,000,000
			ESTIMATED ROUGH-ORDER-OF-MAGNITUDE COST	**$4,000,000**

Notes:
1. Includes Northern Head, East Side, and West Side.
2. Includes use of proprietary binder or weighting agent amendment such as SediMite™ or AquaGate™.
3. Unit prices derived from summary of low- and high-range unit costs presented in Patmont, *et al* (2014).

4. Costs are Rough-Order-of-Magnitude and presented for feasibility-level, comparative purposes only. The project needs to undergo a full design process before numbers can be refined. Consultant makes no warranty, express or implied, that the cost of the work will not vary from these cost values.

Precedent exists for use of GAC for in-situ sediment treatment in an actively used industrial waterway. Puget Sound Shipyard in Bremerton, Washington and Leirvik Sveis Shipyard in Norway, are two examples that are noted in Patmont, *et al* (2014), and USEPA is currently considering a similar approach for the Lower Duwamish Waterway in Seattle, Washington.

POTENTIAL "HYBRID" APPROACH COMBINING TARGETED DREDGING WITH APPLICATION OF ACTIVATED CARBON

Aside from Channel-wide remedial strategies like those discussed above, it may prove beneficial to perform localized dredging at locations where particularly high contaminant levels exist, and combine that with application of GAC to other, less-impacted portions of the Channel.

Targeted dredging would focus specifically on locations with the highest concentrations of DDT. The four areas denoted as targeted "hotspot" dredging on Figures 7 and 8 are areas where DDT concentrations have been measured in excess of 30,000 parts per billion (ppb), and represent a meaningful and reasonable estimation of the worst case areas. The shapes of the dredging extents illustrated on Figures 7 and 8 represent Theissen polygons derived from the arrangement of 2013 sediment data, as compiled and analyzed by Exponent (2015).

Further cleanup benefits can be realized through in-situ treatment by GAC addition, as described in the preceding section of this report. By focusing the GAC application on different selected areas, various remedial end results can be achieved in the Channel. Figures 6 and 7 depict two potential hybrid cleanup alternatives, labeled A and B, in which in-situ treatment with GAC is applied over different target areas based on the amount of DDT exposure reduction achieved in each area. Alternative A represents placement of GAC over 14 Thiessen polygons and subsequent 95% exposure reduction; while Alternative B represents placement of GAC over 18 polygons and subsequent 80% exposure reduction.

Tables 6 and 7 present ROM costs for the two Alternative remedies depicted.

Table 6

ROM Costs for Hybrid Alternative A (Targeted Dredging and GAC Application)

Item	Quantity	Unit	Unit Rate	Cost
Agency Negotiations	1	Lump Sum	$100,000	$100,000
Pre-Design Investigations	1	Lump Sum	$750,000	$750,000
Mobilization and Demobilization	1	Lump Sum	$1,000,000	$1,000,000
Dredging, Sediment Management, and Disposal	20,000[1]	Cubic Yard	$468[2]	$9,360,000
Place GAC product over sediment surface	1.8	Acre	$175,000[3]	$315,000
Environmental Controls	1	Lump Sum	$200,000	$200,000
Long-Term Monitoring and Surveys	5	Episodes	$150,000	$750,000
			Total of Estimated Construction Costs	**$12,480,000**
			Construction Project Add-Ons	
			Technical Design (6%)[4]	$748,800
			Project Management and Overhead (5%)[4]	$624,000
			Construction Management (10%)	$1,248,000
			Total of Estimated Project Costs	**$15,100,800**
			Contingency Factor (35%)	$5,290,000
			TOTAL PROJECTED ROM COST	**$20,390,800**

Notes:
1. Based on area of affected hotspots and anticipated dredge depth, including volume contributed by side slopes and dredging overdepth.
2. Unit price of $468 for dredging and disposal of sediment is from Table 3.
3. Per-acre cost for GAC amendment purchase and placement consistent with costs presented in Table 5.
4. Per USEPA (2000) cost estimating guidance.

ROM = Rough Order-of-Magnitude

Table 7
ROM Costs for Hybrid Alternative B (Targeted Dredging and GAC Application)

Item	Quantity	Unit	Unit Rate	Cost
Agency Negotiations	1	Lump Sum	$100,000	$100,000
Pre-Design Investigations	1	Lump Sum	$750,000	$750,000
Mobilization and Demobilization	1	Lump Sum	$1,000,000	$1,000,000
Dredging, Sediment Management, and Disposal	20,000[1]	Cubic Yard	$468[2]	$9,360,000
Place GAC product over sediment surface	2.9	Acre	$175,000[3]	$507,500
Environmental Controls	1	Lump Sum	$200,000	$200,000
Long-Term Monitoring and Surveys	5	Episodes	$150,000	$750,000
Total of Estimated Construction Costs				**$12,670,000**
Construction Project Add-Ons				
Technical Design (6%)[4]				$760,200
Project Management and Overhead (5%)[4]				$633,500
Construction Management (10%)				$1,267,000
Total of Estimated Project Costs				**$15,330,700**
Contingency Factor (35%)				$5,370,000
TOTAL PROJECTED ROM COST				**$20,700,700**

Notes:
1. Based on area of affected hotspots and anticipated dredge depth, including volume contributed by side slopes and dredging overdepth.
2. Unit price of $468 for dredging and disposal of sediment is from Table 3.
3. Per-acre cost for GAC amendment consistent with costs presented in Table 5.
4. Per USEPA (2000) cost estimating guidance.

ROM = Rough Order-of-Magnitude

It can be seen from these two costs tables that the cost of dredging and disposing of sediment far outweighs the costs of in-situ sediment treatment by GAC application. (This point was established earlier in the discussion of costs presented in Table 3, for sediment dredging, and

Table 5, for GAC placement.) As a result, the costs in Tables 6 and 7 are quite similar in magnitude.

Other hybrid remedy arrangements can be developed based on what SWAC end point is judged to be appropriate; four arrangements developed by Exponent (2015) are included as a set of figures in Attachment B. Despite the varying amounts of area over which GAC is applied, the overall costs would be expected to vary only slightly from those presented above in Tables 6 and 7.

ON-SITE RETENTION OF SEDIMENT WARRANTS FURTHER EVALUATION

Another disposal option, possibly providing future benefits to the site and the community, is to place and permanently confine dredged sediments in a constructed nearshore Confined Disposal Facility (CDF). One concept for a CDF would be to construct an earthen berm across the Channel, place dredged material within the enclosed basin formed by the berm, and then place a clean cap over the material to isolate the contaminants. The end result would be to create usable upland area. This option has been used on several west coast projects, has successfully undergone detailed evaluation by USEPA and other regulatory agencies, and has proven technically effective. It also has the advantage of greatly reducing community impact from truck or rail trips hauling sediment off-site and across the state, as well as associated air quality and greenhouse gas emissions impacts.

The Draft FFS rules out the CDF option in cursory fashion, acknowledging that it "may be an effective disposal option for contaminated sediments," but stating that it requires a "large area," and that "significant administrative or regulatory impediments to implementation are often encountered." These general statements are insufficient to rule out further consideration of the CDF option at this site. There are many cases across the country where administrative and regulatory "obstacles" were successfully overcome, and a CDF was effective in permanently managing and confining contaminated sediments. CDFs are often considered a desirable alternative to hauling dredged sediments long distances across state lines, and can result in usable land space, both of which are positive trade-offs for any regulatory challenges.

A conceptual CDF at the Site, depicted on Figure 9, would be located in the Northern Head of the Lauritzen Channel. Sediment could be placed behind a retaining berm built across the channel, north of the Levin Pier, using sand/gravel fill and armoring rock on the face. This conceptual geometry would result in the filling of approximately 2 acres. Approximately 40,000 cy of sediment could be placed behind the berm to a top elevation of +1 feet MLLW, while the remaining dredged material would be disposed off-site. Another 15,000 cy of sediment would be confined in placed beneath the CDF. By capping the area with clean fill to match surrounding grades, usable land area could be created.

The CDF depicted on Figure 9 would provide cost savings by avoiding transportation and disposal of sediments at a distant or out-of-state upland facility. In addition, the CDF permanently confines existing sediments within its footprint, further reducing the volume of sediment that might require dredging, transportation, and disposal.

Table 8 presents ROM comparative costs for construction of a CDF compared to the amount that would be saved on sediment dredging and off-site disposal. If the additional costs needed to create a fully confined disposal area are less than the amount saved on transport and disposal, then a CDF is a cost-effective remedial option, and Table 8 indicates that the costs could be close to offsetting. (Note one important consideration is the fact that CDF construction would likely require habitat mitigation because it results in a net loss of water area or useful habitat.) Creation of additional usable upland area (approximately 2 acres) at the site may, however, offer a monetary value that will help offset some of the overall CDF costs. In addition, community/environmental impacts from dredging and transport would be significantly lessened due to the fact that less material, or none at all, needs to be hauled off-site.

Table 8

Estimated Construction Costs for Sediment Confined Disposal[1]

Task	Quantity	Unit	Unit Cost	Cost
Additional Equipment Mobilization	1	Lump Sum	$500,000	$500,000
Additional Design and Permitting	1	Lump Sum	$1,500,000	$1,500,000
Dredging of Toe Key for Containment Berm[2]	3,000	CY	$35	$105,000
Construct Berm: Sandy Gravel	9,000	Ton	$30	$270,000
Construct Berm: Armor Rock	7,000	Ton	$50	$350,000
Dredge and place Lauritzen Channel sediment	40,000	CY	$35	$1,400,000
Place clean cap material over confined	30,000	Ton	$25	$750,000
Extend City outfall through CDF to face of berm	1	Lump Sum	$1,000,000	$1,000,000
Base Coarse (6 inches)	3,000	Ton	$30	$90,000
Surfacing Asphalt (4 inch)	87,120	SF	$5	$435,600
Turbidity Curtain for CDF Fill	1	Lump Sum	$100,000	$100,000
Long-term Monitoring	10	event	$100,000	$1,000,000
Mitigation for in-water fill	Uncertain			
			Total	$7,500,600
			Contingency Factor (35%)	$2,625,000
			ESTIMATED ROUGH-ORDER-OF-MAGNITUDE COST[4]	**$10,120,000**
Costs saved, for comparison				
Volume of sediment confined under CDF area,	15,000	CY	$307	$4,600,000
Disposal cost savings for sediment placed in CDF	40,000	CY	$307	$12,300,000
Value of added land	uncertain			

Notes:
1. Conceptual CDF geometry shown on Figure 5
2. Assumes sediment is dredged and relocated into CDF footprint area behind the berm
3. Costs are Rough-Order-of-Magnitude and presented for feasibility-level, comparative purposes only. The project needs to undergo a full design process before numbers can be refined. Consultant makes no warranty, express or implied, that the cost of the work will not vary from these cost values.
4. Does not include cost of mitigation, which would be a significant amount
5. Unit price of $306 per yard is based on the costs presented earlier, for dredging, treatment, transport and disposal; in Table 3.
6. Unit price of $250 per yard is based on costs for sediment treatment, transport, and disposal, as presented earlier in Table 3.

cy = cubic yards
CDF = confined disposal facility
MLLW = mean lower low water
SF = square feet

Altogether, CDF construction on-site is a potentially feasible, cost-effective alternative, which could significantly decrease community impact associated with off-site removal and transport of sediment. As such, it appears to warrant a more thorough review than was presented in the Draft FFS.

It is also worth noting that by combining a CDF at the head of the Channel, with activated carbon amendment throughout the channel or engineered capping near the Levin Pier, there would no longer be any need to haul sediment off site. Any dredged sediment in either combined concept could be contained within the CDF—further reducing costs and environmental/community impacts.

DATA GAPS EXIST IN USEPA'S EVALUATION OF OUTFALLS

One element of the Site cleanup that is carried through to all three alternatives is remedial action at the north end of the channel and the City of Richmond's outfall pipe. It is clear that USEPA has not thoroughly vetted and compared the potential for other conveyances and outfalls to contribute contaminants to the Site.

There are eight known discharge locations (the large municipal outfall, at the head of the channel; five that drain the LRTC Site that are distributed from the head of the channels and along the eastside of the channel; two smaller drains that may be small municipal lines/outfalls), but subtidal discharge locations and those behind the rip-rap are currently uncharacterized. In 2008, nine catch basin samples were collected from the municipal storm drain lines, but none from the LRTC lines.

The most noteworthy catch basin sample was collected from the line that drains the Lauritzen Outfall at the LRTC site boundary (just prior to traversing the LRTC before discharging into the channel), which had DDT concentrations ranging from 38,500 to 52,100 micrograms per kilogram. Given the location of this sample point, it is unclear whether the measured DDT originates from the LRTC site or it is from further up the system/off-site. The Draft FFS recognizes existing knowledge regarding site conveyance systems and outfalls as possible sources of contamination are incomplete, and states:

> *The City of Richmond municipal outfall at the head of the Lauritzen Channel cannot be fully evaluated as an ongoing source of contamination to the Lauritzen Channel until the DDT-contaminated residual sediments within the storm drain system are removed. These sediments will be removed a part of the remedy, and monitoring will be performed to verify that the municipal drains are no longer acting as a DDT transport pathway to the Lauritzen Channel.*

Monitoring the municipal drains after the removal of contaminated sediments in the Channel is inconsistent with a logical sequence of attaining cleanup goals for the site. It would not be prudent for USEPA to analyze remedial alternatives for the Site without first obtaining a better understanding of sources and pathways of contamination. An ongoing source identification problem is potentially fatal to effectively analyzing and weighing remedial alternatives for the Channel.

CONCLUSIONS AND RECOMMENDATIONS

In this technical memorandum, Anchor QEA has used its experience with sediment remediation projects in California and across the nation, to establish that the Draft FFS has significantly underestimated the duration and cost of dredging the Lauritzen Channel and hauling the sediments off-site. All three active alternatives presented in the Draft FFS are largely based on dredging. The Draft FFS should provide a more thorough exploration of the potential advantages and disadvantages of engineered capping and/or placement of an activated carbon layer throughout the channel, confined disposal of sediments within the channel, and various combinations of all three. We have therefore provided a feasibility-level evaluation of these alternatives, and suggest that USEPA incorporate these findings into a finalized version of the FFS which more thoroughly considers alternative approaches to fully dredging the channel. We also recommend that USEPA obtain additional information regarding pathways and sources of contamination before finalizing the Draft FFS and evaluating any remedial alternatives.

The remedial approach which appears to offer the most advantages is the application of activated carbon (mixed with sand or in the form of a proprietary binder product) directly to the sediment surface. This approach minimizes the loss of water depths within the channel, allows for temporary redistribution of the product in the Channel over time, and offers relatively low cost, construction effort, and environmental/community impacts. This approach, along with the others discussed in this memorandum, merit further evaluation by USEPA.

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FIGURES

P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-001 (Location Map).dwg FIG1

AERIAL SOURCE: Google Earth Pro, 2012.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

0 400
Scale in Feet

Apr 21, 2015 4:27pm jbigsby

ANCHOR QEA

Figure 1
Site Location Map
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California

P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-002 (Exist Feat-Bathy).dwg FIG2

Apr 21, 2015 4:30pm jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
SOURCE: Bathymetry contours derived from an EPA CH2M Hill Figure 3-1 2013 Sediment Sample Locations PDF dated 11/12/14.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

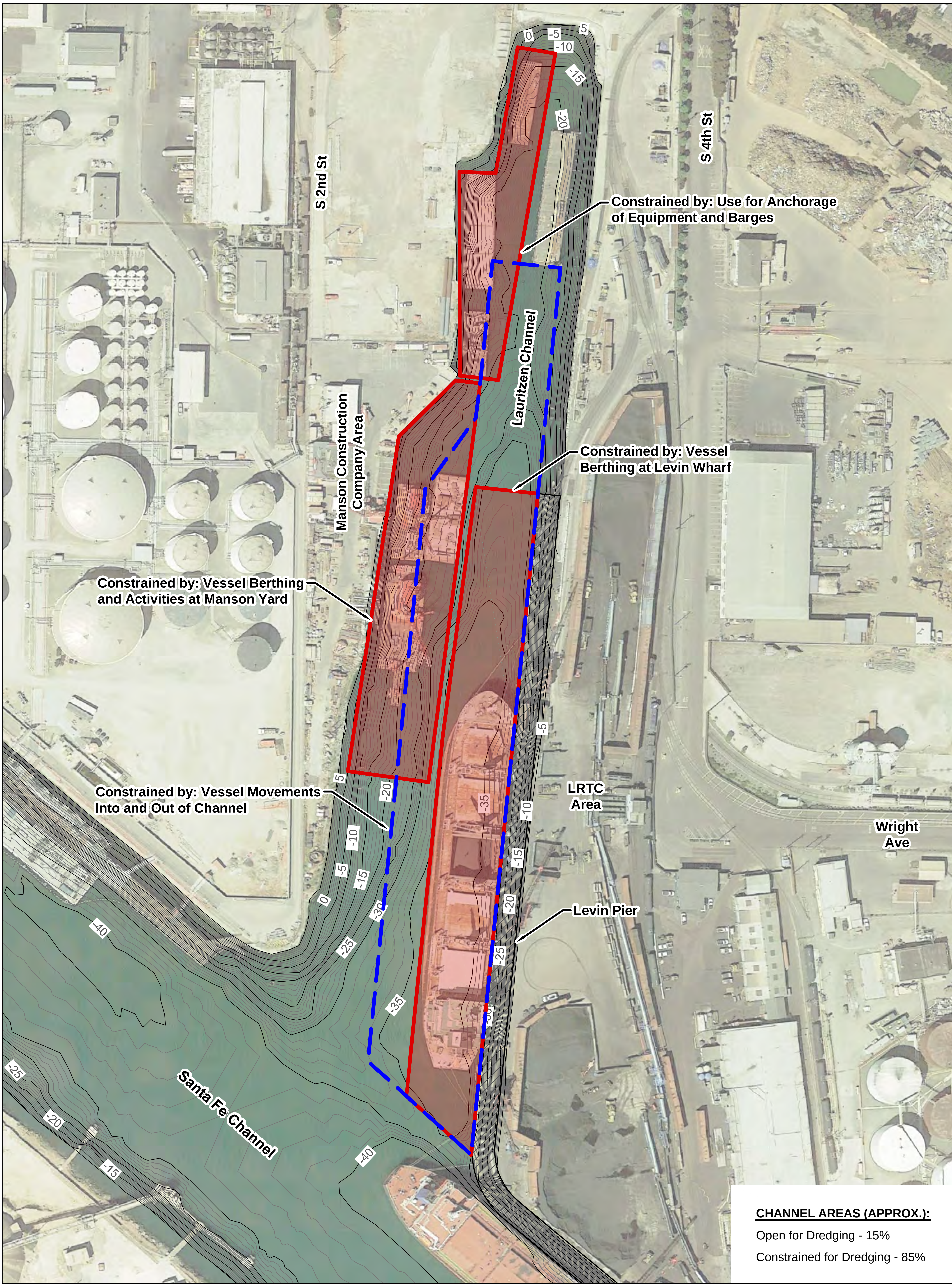
LEGEND:

-40 Existing Bathymetric Contour (5' Intervals from EPA Figure)
Existing Bathymetric Contour (1' Interval Interpolated from 5' Contours)
Levin Pier

0 150
Scale in Feet

ANCHOR QEA

Figure 2
Existing Bathymetry Map
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-003 (Constraints).dwg FIG3

Apr 21, 2015 4:32pm jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
SOURCE: Bathymetry contours derived from an EPA CH2M Hill Figure 3-1 2013 Sediment Sample Locations PDF dated 11/12/14.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

LEGEND:

-40 Existing Bathymetric Contour (5' Intervals from EPA Figure)

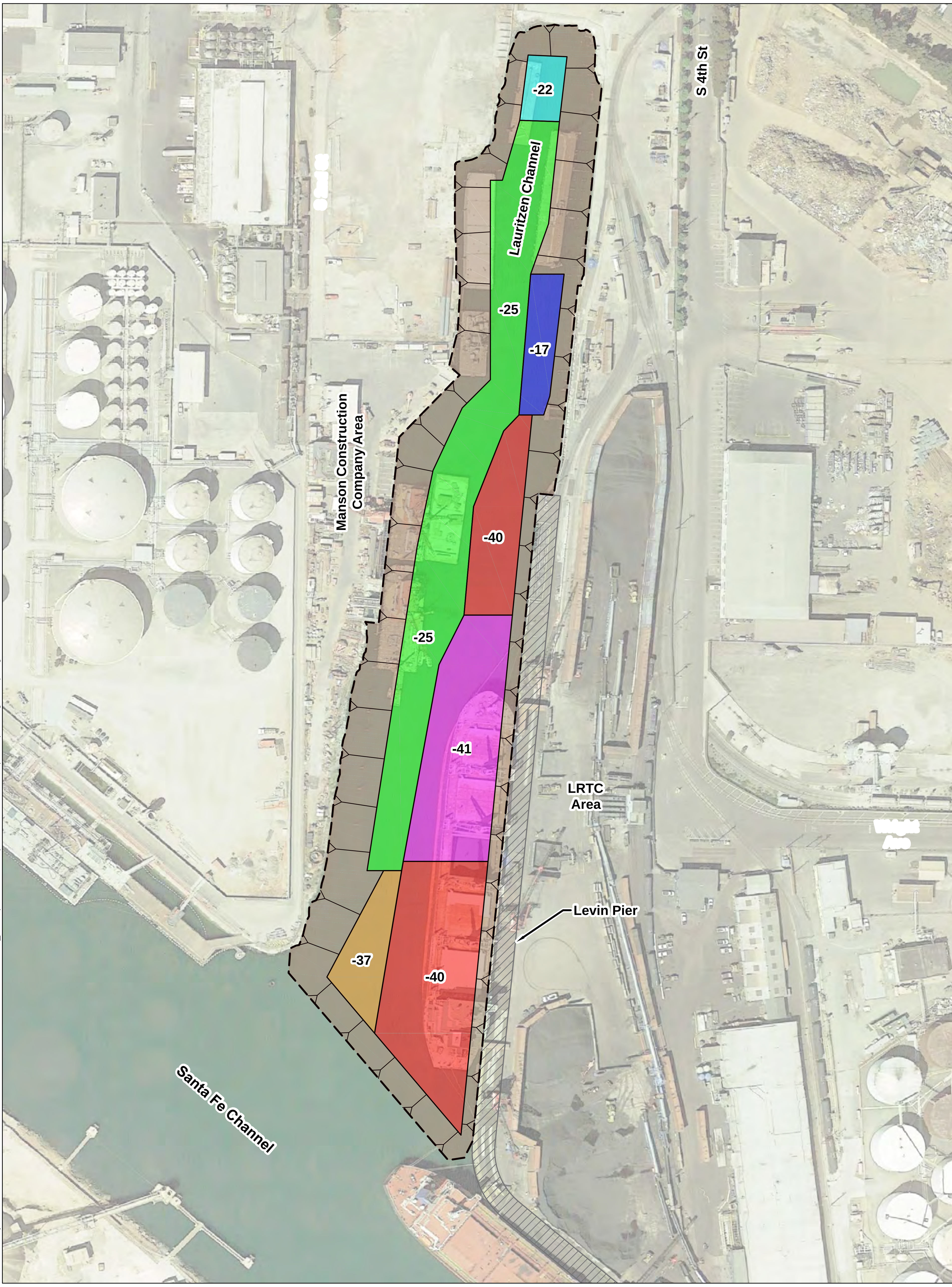
Existing Bathymetric Contour (1' Interval Interpolated from 5' Contours)

Levin Pier

0 150
Scale in Feet

ANCHOR QEA

Figure 3
Areas of Constrained Dredging in Lauritzen Channel
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



P:\CAD\Projects\0754-Latham & Watkins\Montrose Chemical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-004 (Concept Dredge Plan).dwg FIG4

May 20, 2015 9:40am jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

LEGEND:

Levin Pier

Perimeter Slope Daylight Boundary (Conceptual)

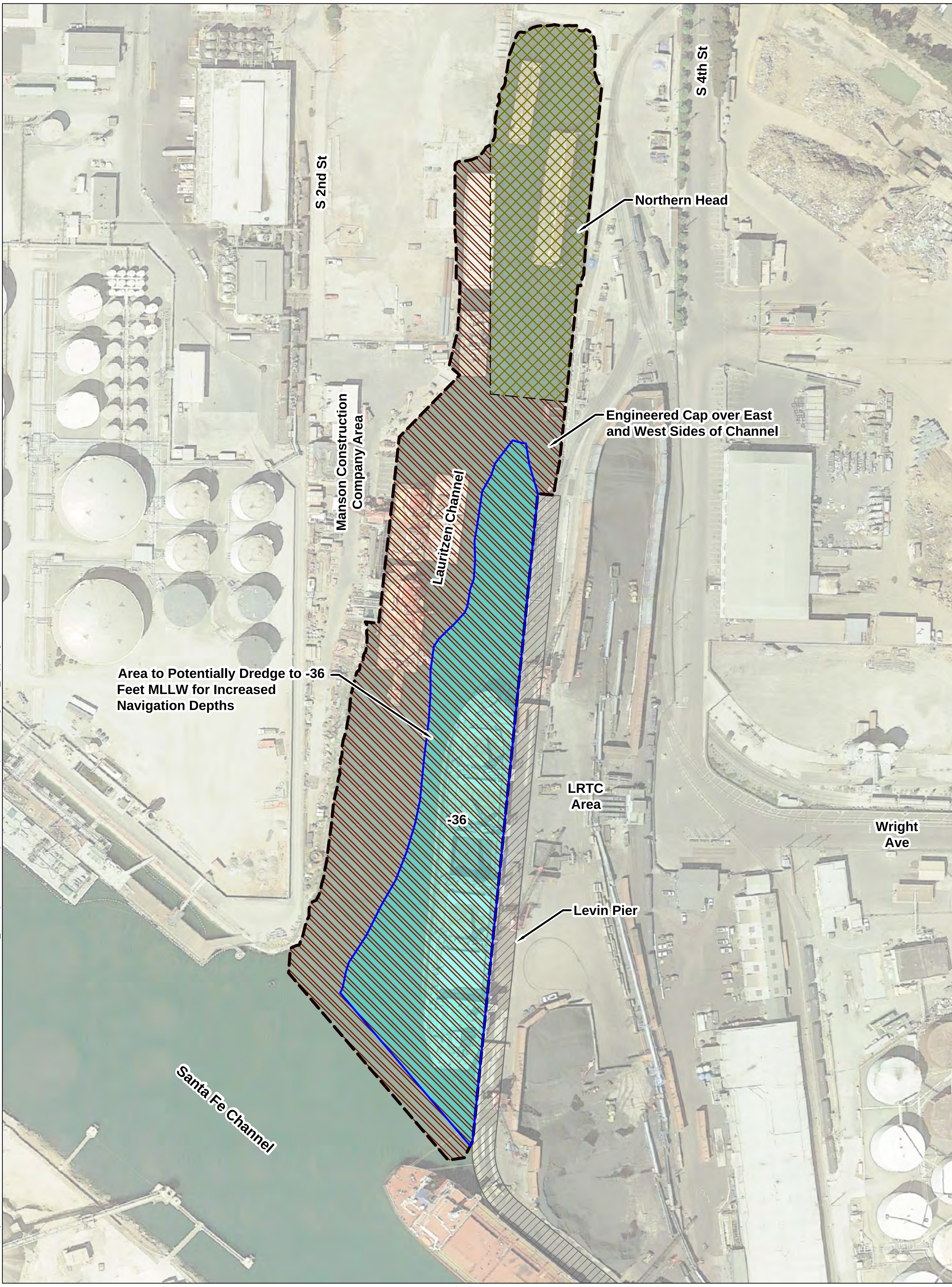
DREDGE PRISM BOUNDARY:

-17' Elevation
-22' Elevation
-25' Elevation
-37' Elevation
-40' Elevation
-41' Elevation

0 150
Scale in Feet

ANCHOR QEA

Figure 4
Conceptual Dredging Plan
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



AERIAL SOURCE: Google Earth Pro, 2012.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

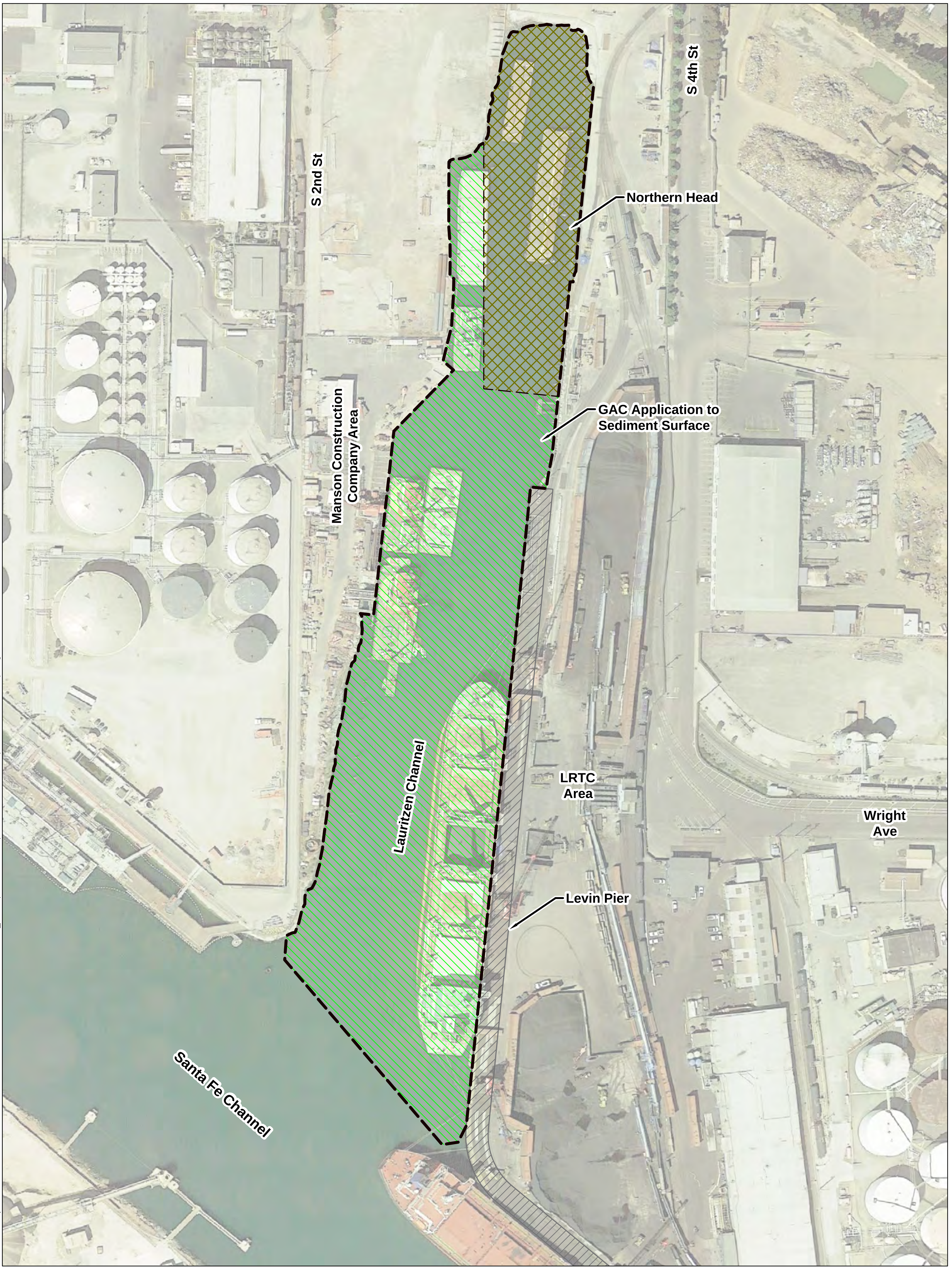
LEGEND:

- Levin Pier
- Perimeter Slope Daylight Boundary (Conceptual)
- Engineered Cap, East and West Sides of Channel
- Northern Head
- Potential Dredge to -36 Feet MLLW

0 150
Scale in Feet

ANCHOR QEA

Figure 5
Potential Dredging in Advance of Engineered Capping
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-006 (GAC-Channel-Wide).dwg FIG6

May 20, 2015 1:40pm jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

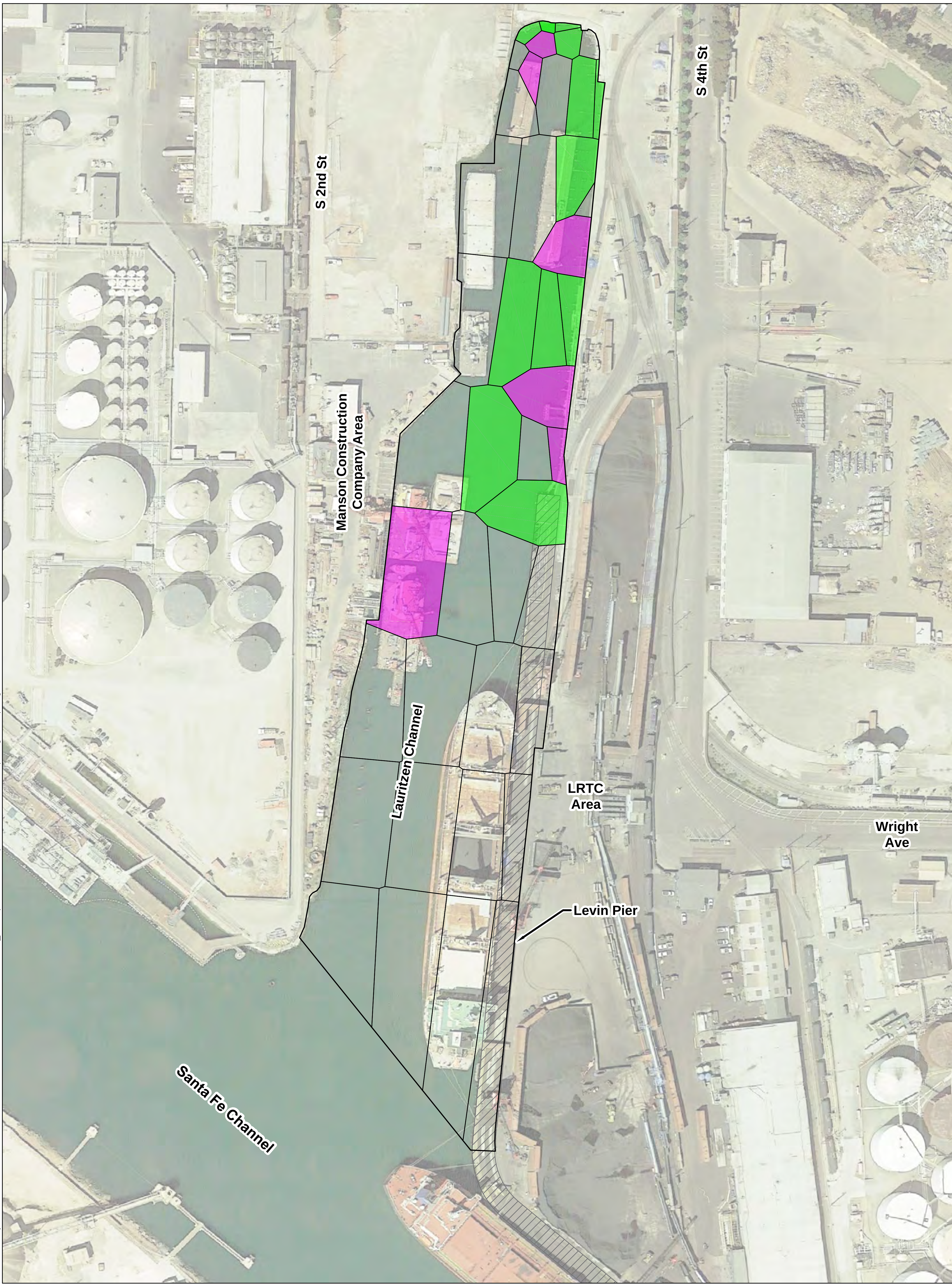
LEGEND:

- Levin Pier
- Perimeter Slope Daylight Boundary (Conceptual)
- GAC Application to Sediment Surface
- Northern Head

0 150
Scale in Feet

ANCHOR QEA

Figure 6
Channel-wide Applilcation of Granular Activated Carbon
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-007 (Dredge-Alt A).dwg FIG7

May 20, 2015 3:20pm jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
SOURCE: Theissen Polygon boundaries from Exponent, 2015.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

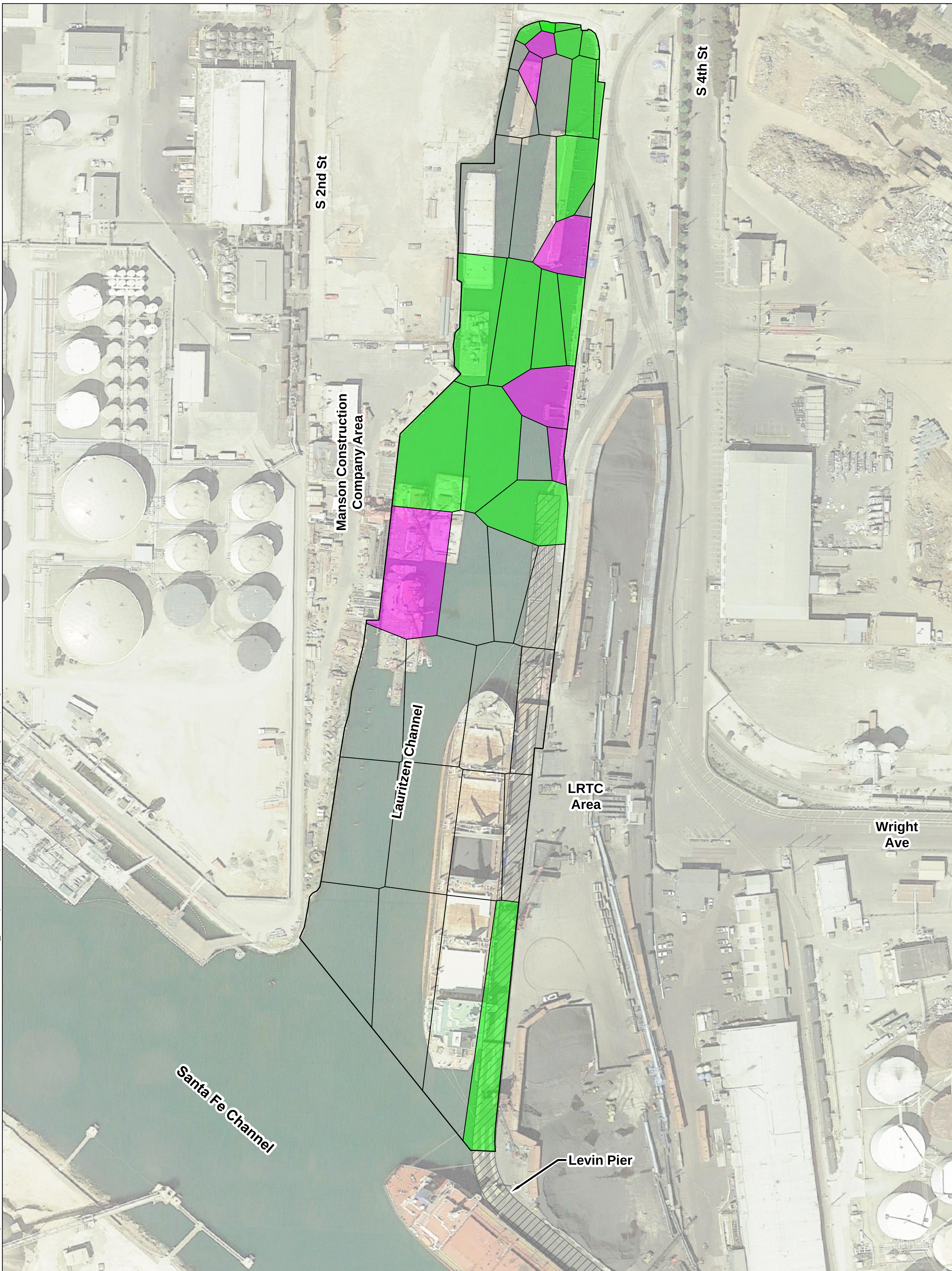
LEGEND:

- Targeted "Hotspot" Dredging
- Localized Application of GAC - Alternative A
- Levin Pier
- Theissen Polygon Boundary

0 150
Scale in Feet

ANCHOR QEA

Figure 7
Dredge Area and GAC Treatment - Alternative A
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California



P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-008 (Dredge-Alt B).dwg FIG8

May 21, 2015 8:38am jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
SOURCE: Theissen Polygon boundaries from Exponent, 2015.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

LEGEND:

- Targeted "Hotspot" Dredging
- Localized Application of GAC - Alternative B
- Levin Pier
- Theissen Polygon Boundary

0 150
Scale in Feet

ANCHOR QEA

Figure 8
Dredge Area and GAC Treatment - Alternative B
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California

P:\CAD\Projects\0754-Latham & Watkins\Montrose Cheimical, United Heckathorn Site_Conceptual Cost Comparison\0754-RP-009 (CDF).dwg FIG9

May 20, 2015 3:39pm jbigsby

AERIAL SOURCE: Google Earth Pro, 2012.
HORIZONTAL DATUM: California State Plane, Zone III, NAD83, U.S. Feet.

LEGEND:

Levin Pier

Perimeter Slope Daylight Boundary (Conceptual)

-40 Dredge Prism Boundary and Elevation in Feet

Usable Filled Land Surface

0 150
Scale in Feet

ANCHOR QEA

Figure 9
Conceptual Confined Disposal Facility (CDF) Area
Engineering Review of Draft FFS
Former United Heckathorn Site, Richmond, California

ATTACHMENT A
PATMONT, ET AL. (2014) ON IN-SITU SEDIMENT TREATMENT USING ACTIVATED CARBON

Integrated Environmental Assessment and Management — Volume 9999, Number 9999—pp. 1–13
Published 2014 SETAC.

In Situ Sediment Treatment Using Activated Carbon: A Demonstrated Sediment Cleanup Technology

Clayton R. Patmont,†‡ Upal Ghosh,§ Paul LaRosa,|| Charles A. Menzie,# Richard G. Luthy,†† Marc S. Greenberg,‡‡ Gerard Cornelissen,§§|||| Espen Eek,§§ John Collins,## John Hull,## Tore Hjartland,††† Edward Glaza,‡‡‡ John Bleiler,§§§ and James Quadrini||||||*

†Anchor QEA, Seattle, Washington, USA
‡Present address: Anchor QEA, 720 Olive Way, Suite 1900, Seattle, WA 98101
§University of Maryland Baltimore County, Baltimore, Maryland, USA
||Anchor QEA, Amesbury, Massachusetts, USA
#Exponent, Alexandria, Virginia, USA
††Stanford University, Stanford, California, USA
‡‡US Environmental Protection Agency, Edison, New Jersey, USA
§§Norwegian Geotechnical Institute, Oslo, Norway
||||Norwegian University of Life Sciences, Ås, Norway
##AquaBlok, Toledo, Ohio, USA
†††BioBlok Solutions, Sandefjord, Norway
‡‡‡Parsons, Syracuse, New York, USA
§§§AECOM, Chelmsford, Massachusetts, USA
||||||Anchor QEA, Woodcliff Lake, New Jersey, USA

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Critical Review

ABSTRACT

This paper reviews general approaches for applying activated carbon (AC) amendments as an in situ sediment treatment remedy. In situ sediment treatment involves targeted placement of amendments using installation options that fall into two general approaches: 1) directly applying a thin layer of amendments (which potentially incorporates weighting or binding materials) to surface sediment, with or without initial mixing; and 2) incorporating amendments into a premixed, blended cover material of clean sand or sediment, which is also applied to the sediment surface. Over the past decade, pilot- or full-scale field sediment treatment projects using AC—globally recognized as one of the most effective sorbents for organic contaminants—were completed or were underway at more than 25 field sites in the United States, Norway, and the Netherlands. Collectively, these field projects (along with numerous laboratory experiments) have demonstrated the efficacy of AC for in situ treatment in a range of contaminated sediment conditions. Results from experimental studies and field applications indicate that in situ sequestration and immobilization treatment of hydrophobic organic compounds using either installation approach can reduce porewater concentrations and biouptake significantly, often becoming more effective over time due to progressive mass transfer. Certain conditions, such as use in unstable sediment environments, should be taken into account to maximize AC effectiveness over long time periods. In situ treatment is generally less disruptive and less expensive than traditional sediment cleanup technologies such as dredging or isolation capping. Proper site-specific balancing of the potential benefits, risks, ecological effects, and costs of in situ treatment technologies (in this case, AC) relative to other sediment cleanup technologies is important to successful full-scale field application. Extensive experimental studies and field trials have shown that when applied correctly, in situ treatment via contaminant sequestration and immobilization using a sorbent material such as AC has progressed from an innovative sediment remediation approach to a proven, reliable technology. *Integr Environ Assess Manag* 2015; 9999:XX–XX. © 2014 The Authors. Published 2014 SETAC.

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* To whom correspondence may be addressed: cpatmont@anchorqea.com
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KEY POINTS

- More than 25 field-scale pilot or full-scale sediment treatment projects performed over the past decade, along with numerous laboratory experiments, have proven the efficacy of in situ sediment treatment using AC to reduce the bioavailability of several hydrophobic organic compounds.
- Controlled placement of AC (accurate and spatially uniform) has been demonstrated using a variety of conventional construction equipment and delivery techniques and in a range of aquatic environments including wetlands.
- In situ sediment treatment using AC has progressed from an innovative remediation approach to a proven, reliable

technology that is ready for full-scale application at a range of sites.

INTRODUCTION

Sediments accumulated on the bottom of a waterbody are recognized as sinks for toxic substances and bioaccumulative chemicals and can be long-term reservoirs for chemicals that can be transferred via the food chain to invertebrates and fish (USEPA 2005). Establishing effective methods to reduce the ecological and human health risks contaminated sediment poses has been a regulatory priority in North America, Europe, and elsewhere since the 1970 s. Indeed, demonstrating risk reduction that is convincing to all stakeholders using traditional dredging and isolation capping approaches has been challenging (NRC 2007; Bridges et al. 2010). Although traditional approaches will continue to be an integral part of sediment cleanup remedies (e.g., when contaminated sediments are present in unstable environments), new remediation approaches are needed to either supplement or provide alternatives to existing methods.

In situ sediment treatment via contaminant sequestration and immobilization generally involves applying treatment amendments onto or into surface sediments (Luthy and Ghosh 2006; Supplemental Figure S1). This paper reviews the considerable advances in engineering approaches used to apply activated carbon (AC)-based treatment amendments in situ; summarizes field-scale demonstration pilots and full-scale applications performed through 2013; and describes lessons learned on the most promising application options. This paper also discusses the need for a balanced consideration of the potential benefits, ecological effects, and costs of in situ treatment using AC relative to other sediment cleanup technologies. The results of this work aim to identify a common set of features from engineering, chemistry, and ecology that could help guide and advance the use of in AC-based in situ sediment treatment in future sediment remediation projects.

TREATMENT AMENDMENTS AND MECHANISMS

Beginning in the early 2000s, encouraging results from laboratory tests and carefully controlled, small-scale field studies generated considerable interest in remediating, or managing, contaminated sediments in situ. Mechanisms to do so mainly suggested sorptive treatment amendments such as AC, organoclay, apatite, biochar, coke, zeolites, and zero valent iron (USEPA 2013a). Three of these amendments—AC, organoclay, and apatite—have been identified as particularly promising sorptive amendments for in situ sediment remediation (USEPA 2013b). Of these, AC has been used more widely in laboratory experiments and field-scale applications to control dissolved hydrophobic organic compounds (HOCs). This is largely because AC has been used successfully for decades as a stable treatment medium for water, wastewater, and air, and because early testing of sediment treatment with AC showed positive results.

Laboratory testing and field-scale applications of AC have demonstrated its effectiveness in reducing HOC bioavailability. Both natural and anthropogenic black carbonaceous particles in sediments, including soot, coal, and charcoal strongly bind HOCs, and the presence of these particles in sediments has been demonstrated to reduce biouptake and exposure substantially (Gustafsson et al. 1997; Cornelissen et al. 2005). Using engineered black carbons such as AC augments the native sequestration capacity of sediments, resulting in reduced in situ bioavailability of HOCs. When AC is applied at optimal, site-specific doses (often similar to the native organic carbon content of sediment), the porewater concentrations and bioavailability of HOCs can be reduced between 70% and 99%. Furthermore, AC-moderated HOC sequestration often becomes more effective over time due to progressive mass transfer (Millward et al. 2005; Zimmerman et al. 2005; Werner et al. 2006; Sun et al. 2009; Ghosh et al. 2011; Cho et al. 2012).

Given these promising results, in situ sediment treatment involving the use of AC amendments is receiving increased attention among scientists, engineers, and regulatory agencies seeking to expand the list of remedial technologies and address documented or perceived limitations associated with traditional sediment remediation technologies. Based on the authors' review, AC is now the most widely used in situ sediment sequestration and immobilization amendment worldwide.

A previous review of the in situ AC remediation approach (Ghosh et al. 2011) reported the results of laboratory studies and early pilot-scale trials, summarized treatment mechanisms, highlighted promising opportunities to use in situ amendments to reduce contaminant exposure risks, and identified potential barriers for using this innovative technology. Another critical review by Janssen and Beckingham (2013) summarized the dependence of HOC bioaccumulation on AC dose and particle size, as well as the potential impacts of AC amendments on benthic communities (e.g., higher AC dose and smaller AC particle size further reduce bioaccumulation of HOCs but may induce stress in some organisms). This paper builds on these earlier reviews, focusing on design and implementation approaches involving the use of AC for in situ sediment treatment and summarizing key lessons learned.

DEMONSTRATING EFFICACY IN THE FIELD

Until recently, a primary challenge for full-scale in situ treatment remedies has been that most experience has emerged from laboratory and limited field pilot studies. Through 2013, however, more than 25 field-scale demonstrations or full-scale projects spanning a range of environmental conditions were completed or underway in the United States, Norway, and the Netherlands (Table 1 and Figure 1).

Among the more than 25 projects, field demonstrations in the lower Grasse River (Massena, NY, USA) and upper Canal Creek (Aberdeen, MD, USA) included the most comprehensive assessments and available documentation of the longer-term efficacy of the in situ AC remediation approach, although similar results have been reported for many of the other field projects. For this reason, the lower Grasse River and upper Canal Creek field demonstrations receive the greatest attention here, as summarized below.

Demonstration in lower Grasse River, Massena, New York

An AC pilot demonstration was conducted in the lower Grasse River as part of a program designed to evaluate available sediment cleanup options for the site. The demonstration study evaluated the effectiveness of AC as a means to sequester sediment polychlorinated biphenyls (PCBs) and reduce flux from sediments and uptake by biota.

The project began with laboratory studies and land-based equipment testing, and continued with field-scale testing of alternative placement methods. It culminated in a 2006 field demonstration of the most promising AC application and mixing methods to a 0.2-hectare pilot area of silt and fine sand sediments

Table?1. In situ sediment treatment using carbon-based sorbents (mainly AC): Summary of field-scale pilot demonstrations or full-scale projects

Site number (see Figure 1)	Year(s)	Location	Contaminant(s)	Application area (hectares)	Carbon-based amendment(s)	Delivery method(s)	Average water depth during delivery (m)	Enhancement(s)	Application equipment	Primary reference(s)
1	2004	Anacostia River, Washington, DC	PAHs	0.2	Coke Breeze	Geotextile mat	8	Armored cap	Crane	McDonough et?al. (2007)
2	2004, 2006	Hunters Point, San Francisco, CA	PCBs, PAHs	0.01	AC (slurry)	Direct placement	<1	Mechanical mixing (some areas)	Aquamog, slurry injection	Cho et?al. (2009 and 2012)
3	2006	Grasse River, Massena, NY	PCBs	0.2	AC (slurry)	Direct placement	5	Mechanical mixing (some areas)	Tine sled injection, tiller (with and without mixing)	Beckingham et?al. (2011); Alcoa (2007)
4	2006, 2008	Trondheim Harbor, Norway	PAHs, PCBs	0.1	AC (slurry)	Blended cover, direct placement	5	Armored cap (some areas)	Tremie, agricultural spreader	Cornelissen et?al. (2011)
5	2006	Spokane River, Spokane, WA	PCBs	1	Bituminous Coal Fines (slurry)	Direct placement	5	Armored cap	Mechanical bucket	Anchor QEA (2007 and 2009)
6	2009	De Veenkampen, Netherlands	Clean Sediment	<0.01	AC (slurry)	Direct placement	1	None	Laboratory rollerbank	Kupryianchyk et?al. (2012)
7	2009	Greenlandsfjords, Norway	Dioxins/Furans	5	AC (slurry)	Blended cover	30/100	None	Tremie from hopper dredge	Cornelissen et?al. (2012)
8	2009	Bailey Creek, Fort Eustis, VA	PCBs	0.03	AC (SediMite®)	Direct placement	1	None	Pneumatic spreader	Ghosh and Menzie (2012)
9	2010	Fiskerstrand Wharf, Ålesund, Norway	TBT	0.2	AC (slurry)	Blended cover	40	None	Tremie with biokalk	Eek and Schaanning (2012)
10	2010	Tittabawassee River, Midland, MI	Dioxins/Furans	0.1	AC (AquaGate™), Biochar	Blended cover	<1	None	Agricultural disc	Chai et?al. (2013)
11	2011	Upper Canal Creek, Aberdeen, MD	PCBs, Mercury	1	AC (SediMite®, AquaGate™, slurry)	Direct placement	<1	None	Pneumatic spreader, bark blower, hydroseeder	Bleiler et?al. (2013); Menzie et?al. (2014)
12	2011	Lower Canal Creek, Aberdeen, MD	Mercury, PCBs	0.04	AC (SediMite®)	Direct placement	1	None	Agricultural spreader	Menzie et?al. (2014)
13	2011 to 2016	Onondaga Lake, Syracuse, NY	Various Organic Chemicals	110	AC (slurry)	Blended cover	5	Armored cap	Hydraulic spreader	Parsons and Anchor QEA (2012)

(Continued)

Table 1. (*Continued*)

Site number (see Figure 1)	Year(s)	Location	Contaminant(s)	Application area (hectares)	Carbon-based amendment(s)	Delivery method(s)	Average water depth during delivery (m)	Enhancement(s)	Application equipment	Primary reference(s)
14	2011	South River, Waynesboro, VA	Mercury	0.02	Biochar (Cowboy Charcoal®)	Direct placement	<1	None	Pneumatic spreader	DuPont (2013)
15	2011	Sandefjord Harbor, Norway	PCBs, TBT, PAHs	0.02	AC (BioBlok[a])	Direct placement	30	None	Mechanical bucket	Lundh et?al. (2013)
16	2011	Kirkebukten, Bergen Harbor, Norway	PCBs, TBT	0.7	AC (BioBlok[a])	Direct placement	30	Armored cap (some areas)	Mechanical bucket	Hjartland et? al. (2013)
17	2012	Leirvik Sveis Shipyard, Sandefjord, Norway	PCBs, TBT, Various Metals	0.9	AC (BioBlok[a])	Direct placement	30	Armored cap (some areas)	Hydraulic spreader (up to 30-degree slopes)	Lundh et?al. (2013)
18	2012	Naudodden, Farsund, Norway	PCBs, PAHs, TBT, Various Metals	0.4	AC (BioBlok[a])	Direct placement	30	Armored cap, habitat layer	Mechanical bucket	Lundh et?al. (2013)
19	2012	Berry's Creek, East Rutherford, NJ	Mercury, PCBs	0.01	AC (SediMite®, granular)	Blended cover, direct placement	<1	None	Pneumatic spreader	USEPA (2013c)
20	2012	Puget Sound Shipyard, Bremerton, WA	PCBs, Mercury	0.2	AC (AquaGate™)	Direct placement	15	Armored cap	Telebelt® (under-pier)	Johnston et? al. (2013)
21	2012	Custom Plywood, Anacortes, WA	Dioxins/Furans	0.02	AC (SediMite®)	Blended cover, direct placement	8	None	Agricultural spreader	WDOE (2012)
22	2012	Duwamish Slip 4, Seattle, WA	PCBs	1	AC (slurry)	Blended cover	4	Armored cap	Mechanical bucket	City of Seattle (2012)
23	2013	Mirror Lake, Dover, DE	PCBs, Mercury	2	AC (SediMite®)	Direct placement	1	None	Telebelt® and air horn	DNREC (2013)
24	2013	Passaic River Mile 10.9, Newark, NJ	Dioxin/Furans, PCBs	2	AC (AquaGate™)	Blended cover	1	Armored cap	Telebelt®	In preparation
25	2013	Little Creek, Norfolk, VA	PCBs, various metals	1	AC (AquaGate™)	Direct placement	1	None	Pneumatic spreader (under-pier)	In preparation

AC, activated carbon; PAH, polynuclear aromatic hydrocarbon; PCB, polychlorinated biphenyl; TBT, tributyltin.
[a]BioBlok is licensed by AquaBlok®.

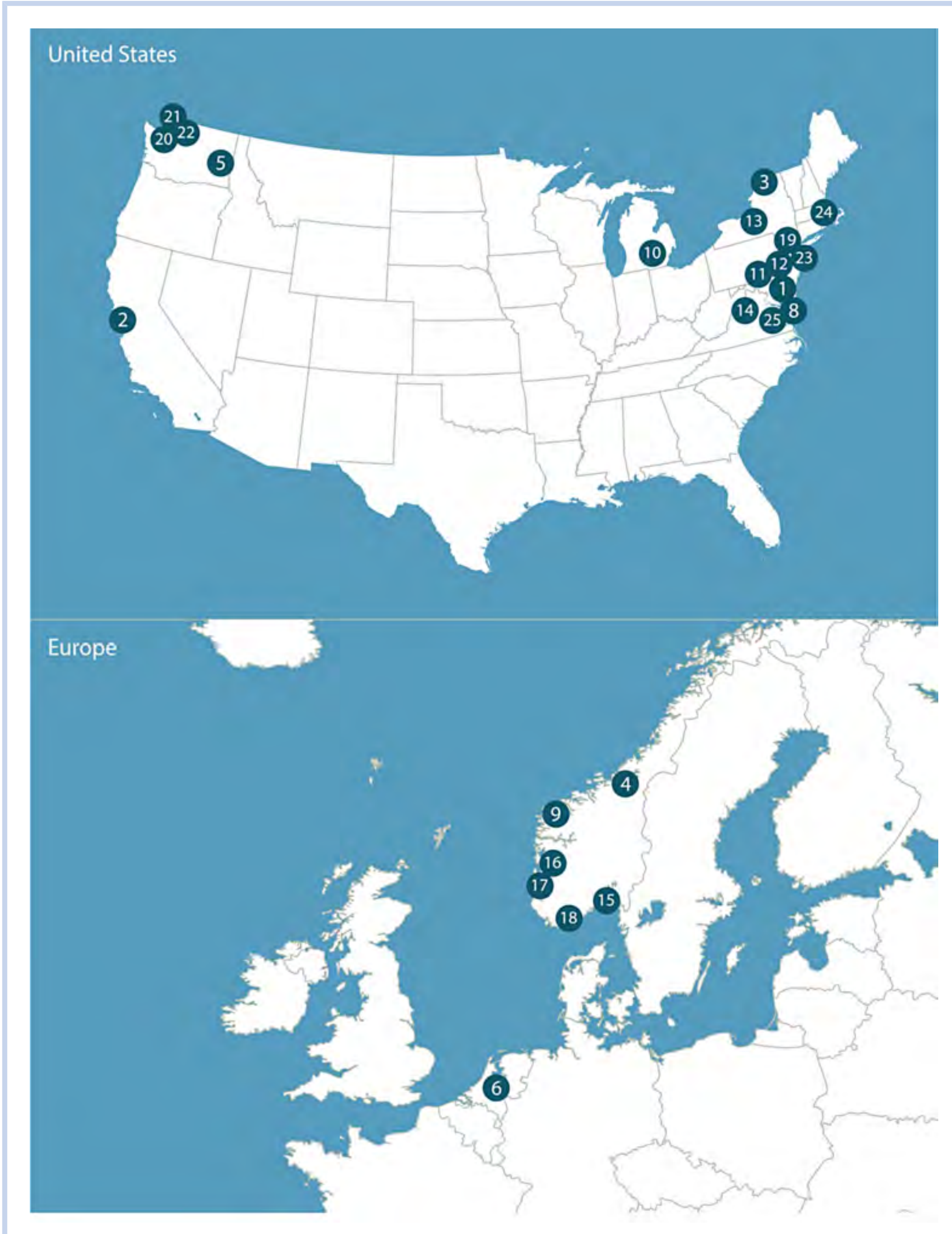


Figure 1. In situ sediment treatment field application sites (numbers refer to sites listed in Table 1).

at average water depths of approximately 5 meters (Alcoa 2007; Beckingham and Ghosh 2011).

The following application techniques were implemented in the Grasse River (Supplemental Figure S2):

- Applying (spraying) an AC slurry onto the submerged sediment surface and then mixing the material into near-surface sediments using a rototiller-type mechanical mixing unit (tiller)
- Injecting an AC slurry directly into near-surface sediments using a tine sled device (tine sled)
- Applying (spraying) an AC slurry onto the sediment surface within a temporary shroud enclosure, with no sediment mixing

All three application techniques successfully delivered the AC slurry onto or into surface sediments, and no detectable losses of AC to the water column or water quality impacts (e.g., turbidity monitored using instrumentation) were observed during placement (Alcoa 2007). A chemical oxidation method developed by Grossman and Ghosh (2009) was used to quantitatively confirm AC doses delivered onto or into sediment. This particular analytical method was used because typical total organic carbon and thermal (375 °C) oxidation methods were found to be imprecise and inaccurate, respectively, for AC analysis in sediment. Spraying the slurry onto the sediment successfully delivered AC to the sediment surface, and both the tiller with mixing and the tine sled applied all of the delivered AC into the 0- to 15-cm sediment layer. The tine sled application achieved more spatially (laterally) uniform doses, with an average AC concentration delivered to the 0- to 15-cm sediment layer of approximately $6.1 \pm 0.8\%$ AC (dry wt; ± 1 standard error around the mean based on core and surface grab sample data). This target (and applied) dose was approximately $1.5\times$ the native organic carbon content of the lower Grasse River. Cost comparisons of the different placement techniques indicate the tine sled unit would be a more cost-effective delivery method under full-scale deployment.

Detailed post-construction monitoring of the AC pilot area was performed in 2007, 2008, and 2009 (Beckingham and Ghosh 2011). Key findings are summarized below:

- AC addition decreased sediment porewater PCB concentrations, and reductions improved during the 3-year, post-placement monitoring period. Greater than 99% reductions in PCB aqueous equilibrium concentrations were observed during the third year of post-placement monitoring in plots where the AC dose in the 0- to 15-cm layer was 4% or greater (Figure 2), effectively demonstrating that PCB flux from sediments to surface water was almost completely contained.
- AC addition decreased PCB bioavailability as measured by in situ and ex situ bioaccumulation testing (using *Lumbriculus variegatus*). The overall decrease improved during the 3-year, post-placement monitoring period, with greater than 90% reductions observed during the third year of post-placement monitoring in plots where the AC dose in the 0- to 15-cm layer was greater than 4% (Figure 2).
- Benthic recolonization occurred rapidly after application and no changes to the benthic community structure or number of individuals were observed in AC amendment plots relative to background (Beckingham et al. 2013).
- In laboratory studies using site sediment, aquatic plants grew at a moderately reduced rate (approximately 25% less than controls) in sediment amended with a dose of greater than 5% AC. The reduced growth rate was likely attributable to nutrient dilution of the sediment (Beckingham et al. 2013).
- Although other project data (not shown) indicated the AC amendment slightly increased the erosion potential of sediments (although within the range of historical data for native sediments), all of the delivered AC remained in the sediments throughout the 3-year, post-placement monitoring period.
- Up to several centimeters of relatively clean, newly deposited sediment accumulated on the sediment surface in the pilot area over the 3-year, post-placement monitoring period. Passive sampling measurements revealed a downward flux of freely dissolved PCBs from the overlying water column into the AC amended sediments throughout the post-construction monitoring period. This suggested that the placed AC will continue to reduce PCB flux from sediments in the long term.

Demonstrations in upper Canal Creek, Aberdeen Proving Ground, Maryland

Two interrelated, pilot-scale, field demonstration projects were performed in 2011 to evaluate AC amendment additions to hydric soils at a tidal estuarine wetland in upper Canal Creek, at the Aberdeen Proving Ground, Maryland. (A third, separate treatment study was also carried out in the channelized portion of lower Canal Creek, but those results are only described minimally here.)

The first demonstration pilot (Menzie et al. 2014) evaluated in situ treatment with SediMite® pellets, a proprietary system for delivering powdered AC treatment materials with a weighting agent and an inert binder (Ghosh and Menzie 2010 2012). The second demonstration pilot (Bleiler et al. 2013) evaluated two different powdered AC-bearing treatment materials: AquaGate + PAC™ (AquaGate) and a slurry containing AC. The proprietary AquaGate product typically includes a dense aggregate core, along with clay-sized materials, polymers, and powdered AC additives. For both field demonstrations and all AC-bearing materials, the objective was to reduce PCB exposure to invertebrates living on or within surface sediments of the wetland area and thus reduce exposure to wildlife that might feed on these invertebrates.

All three AC-containing treatment materials for these pilot projects were applied onto the surface of the wetland and creek sediments during seasonal and tidal conditions with little or no overlying water. A total of 20 plots (each 8×78 meters) were

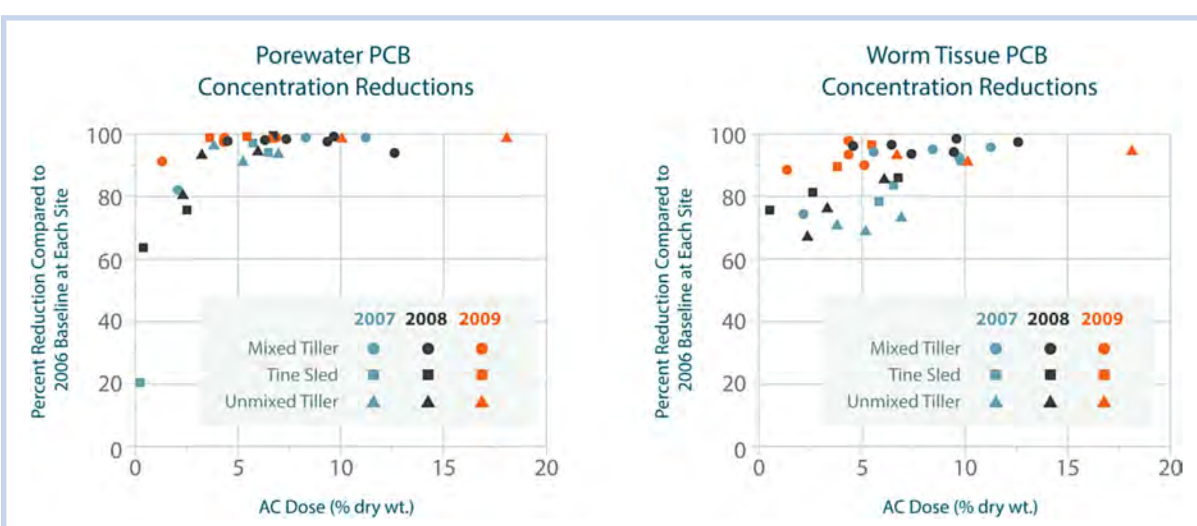


Figure 2. Reductions in porewater and worm tissue PCB concentrations at lower Grasse River, NY.

used for the demonstration projects; sampling was conducted prior to application and at 6 and 10 months following application. Performance measurements used in one or both of the pilot projects included porewater and macroinvertebrate tissue PCB concentrations; phytotoxicity bioassays; ecological community abundance, diversity, and growth surveys; and nutrient uptake studies. Treatment efficacy was evaluated by comparing pre- versus post-treatment metrics and by evaluating treated plots relative to control (no action) and conventional sand cap plots.

The three treatment materials—SediMite®, AquaGate, and AC in a slurry—were applied using a pneumatic spreader, a bark blower, and a hydroseeder, respectively (Supplemental Figure S3). Figure S3 also shows a barge-mounted agricultural spreader that was used to demonstrate delivery of SediMite® to a portion of lower Canal Creek.

For both field demonstrations and all AC-bearing materials, the treatment goal was to achieve a 3% to 7% (dry wt) AC concentration in wetland surface sediment, which was operationally defined as the upper 10 cm (SediMite® studies) and 15 cm (AquaGate and slurry studies). Because the materials contained different amounts of AC, the applications differed in target thickness on the wetland surface. SediMite® contains approximately 50% AC by dry weight, so the target dose of 5% in the top 10 cm of sediment resulted in a target amendment layer thickness of roughly 0.7 cm. In contrast, AquaGate contained a coating of 5% powdered AC and was thus applied as a thicker 3-cm to 5-cm target layer over the sediment. The slurry system delivered roughly 0.2 cm to 0.5 cm of concentrated AC on the surface of the marsh. All of the treatments relied on natural processes (bioturbation, sediment deposition, and other physical processes) to mix AC placed onto the sediment surface into the wetland and creek sediment over time (see post-construction monitoring discussion below).

The AC amendments were applied effectively onto wetland and creek sediments in all of the applications. Measurements made over time indicated that close to 100% of the AC was retained within the plots, but vertical mixing into native wetland sediments via natural processes was slower than originally anticipated. As a result of low bioturbation rates, AC applied in more concentrated forms (i.e., as SediMite® and as AC in a slurry) remained at concentrations greater than the target dose of 5% in the upper 2 cm of the wetland sediment layer 10 months following application (Supplemental Figure S4). During the 10-month, post-application monitoring period, AC was incorporated into the biologically active zone largely from localized root elongation processes (Bleiler et al. 2013). Based on the two post-application monitoring rounds, approximately 60% of the recovered AC was found in the top 2 cm of sediment, whereas the remaining 40% penetrated mostly in the 2- to 5-cm depth interval. It is expected that further incorporation of the AC into the deeper layers of sediment will occur slowly over time via natural mixing processes and deposition of new sediment and organic matter.

The effectiveness of the AC amendments applied to the upper Canal Creek wetlands was assessed by measuring reductions in PCB concentrations in porewater (in situ measurements) and macroinvertebrate tissue (ex situ bioaccumulation testing). PCB concentrations exhibited a large spatial variability (1 order of magnitude) and vertical variability (up to 2 orders of magnitude within a sediment depth of 20 cm) in sediments across the plots, which was a site condition before the AC was applied. This finding posed some challenges in interpreting data and was therefore taken into account when evaluating other metrics. The findings of the upper Canal Creek demonstration pilot are reported in detail in Menzie et al. (2014) and Bleiler et al. (2013).

Regardless of the above challenges, all AC-treated wetland plots showed reduced PCB bioavailability as measured by reductions in both benthic organism tissue and porewater concentrations during the post-application monitoring period. In addition, no significant phytotoxicity or changes in species abundance, richness or diversity, vegetative cover, or shoot weight or length were observed between the AC treatment and control plots. Furthermore, plant nutrient uptake in the AC treatment plots was not significantly lower than control plots. Although the overall findings of these pilot projects suggest that adding AC can sequester PCBs in wetland sediments, more monitoring will take place given the slow mixing of the placed AC into the underlying wetland and creek sediments.

The lower Grasse River and upper Canal Creek projects, along with the other field-scale projects summarized in Table 1, collectively demonstrate the efficacy of full-scale in situ sediment sequestration and immobilization treatment technologies. Such efforts reduce the bioavailability and mobility of several HOC and other contaminants, including PCBs, polynuclear aromatic hydrocarbons, dioxins and furans, tributyltin, methylmercury, and similar chemicals. Results from these field applications indicate that in situ treatment of contaminants can reduce risks rapidly by addressing key exposures (e.g., bioaccumulation in invertebrates), often becoming more effective over time due to progressive mass transfer.

APPLICATION METHODS AND EXAMPLES

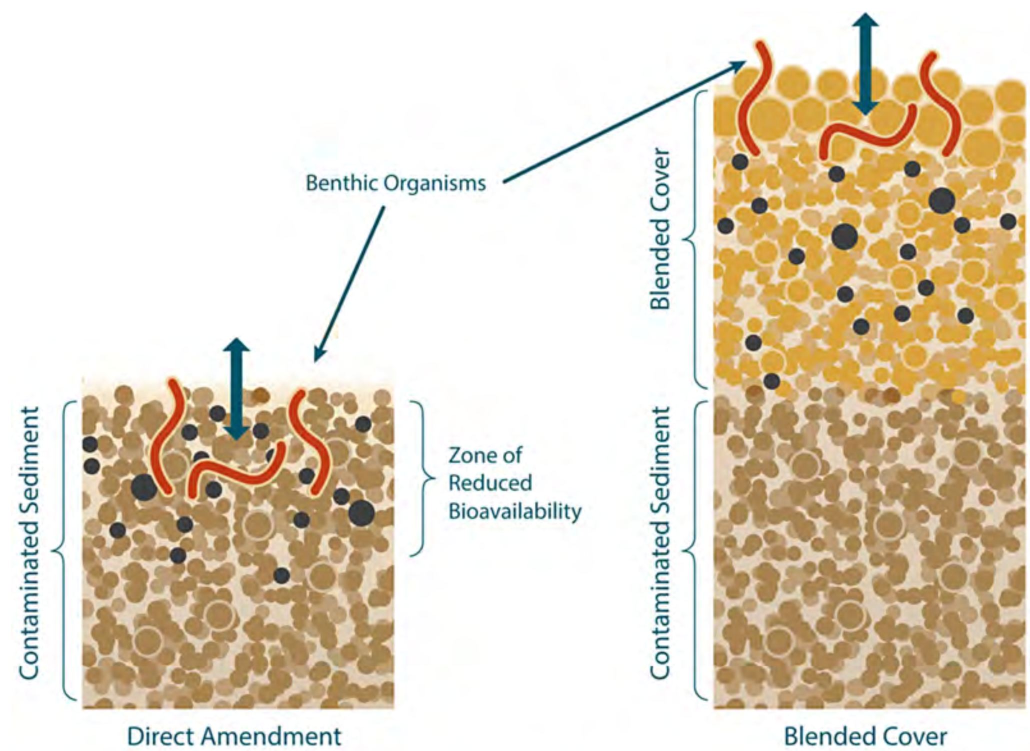
The AC application projects summarized in Table 1 involved placing amendments using several options that fall into two broad categories (Figure 3):

1) Direct application of a thin layer of sorptive, carbon-based amendments (which potentially incorporates weighting or binding materials) onto the surface sediment, with or without initial mixing
2) Incorporating amendments into a pre-mixed, blended cover material of clean sand or sediment, which is also applied onto the sediment surface

Although these approaches have several differences, the ultimate goal of both is to reduce exposure of benthic organisms to HOCs in sediment and reduce HOC flux from sediment into water (Figure 3). Under either approach, the applied AC may mix eventually throughout the biologically active layer via bioturbation. Application methods are described further in the next sections.

Direct application method

Using this approach, the bioavailability of HOCs in surface sediments is reduced by directly applying a strong carbon-based sorbent such as AC. At the lower Grasse River, upper Canal Creek, and many other field demonstration or full-scale projects (Table 1), AC amendment was applied successfully using several methods with or without mixing, weighting agents, inert binders, or other proprietary systems. The specific application method was optimized to site-specific conditions.



Sediment amendment involves integration of a sorbent like AC into the surficial bioactive zone of sediments (shown on the left) while blended cover involves placement of a new layer of cover materials (typically a relatively thin layer of clean sand or sediment) that includes a sorbent like AC either dispersed within (shown in right) or placed as a discrete layer as part of a multi-layer cover. Although these approaches have several differences, the ultimate goal of both approaches is to reduce exposure of benthic organisms to HOCs in sediments and also to reduce HOC flux from sediment into water.

Figure 3. Direct amendment versus blended cover application methods for in situ sorbent application.

Adding weighting agents or inert binders can often improve the placement accuracy of finer-grained AC materials.

When the amendment introduced consists primarily of the sorbent, the direct application approach introduces minimal new material (an advantage), with little or no change in bathymetry or ecological habitat including the sediment's physical and mineralogical characteristics. Applying amendment to sediment surfaces also allows for some capacity to treat new contaminated sediments that may be deposited after constructing the remedy. This approach may have particular advantages at ecologically sensitive sites, where maintaining water depth is critical, and also where the potential for erosion is low.

The Delaware Department of Natural Resources and Environmental Control conceived and funded the first full-scale example of direct placement of AC in the United States, which was implemented in Mirror Lake, a reservoir on the St. Jones River in Dover, Delaware (Table 1; Site 23). The sediment cleanup remedy at this site aimed to enhance the sorption capacity of native sediments in the lake, such that PCB bioavailability to the food chain is reduced without greatly altering the existing sediment bed. The remedy included placing SediMite® over an approximate 2-hectare area in the lake and river, along with integrated habitat restoration (DNREC 2013).

Placing AC at Mirror Lake was performed in the fall of 2013 using two application methods (Supplemental Figure S5): a Telebelt® application for the most accessible parts of the lake and an air horn device to pneumatically deliver SediMite® from a boat and along nearshore areas. Heavy equipment could not be deployed in the lake due to shallow water depth (averaging roughly 1 meter), as well as soft bottom sediments. The SediMite® application was completed safely in approximately 2 weeks. The target (and measured) thickness of the applied SediMite® material was approximately 0.7 cm, with the material expected to integrate naturally into the surficial sediment over time. Grab samples (13 stations) were collected from the top 10 cm of sediment in the lake 2 weeks after application to measure AC based on a method described in Grossman and Ghosh (2009). Applying SediMite® achieved an average AC dose of $4.3 \pm 1.6\%$ (Supplemental Figure S6).

Blended cover application method

The blended cover application method is a variation of the enhanced natural recovery remedy described by the US Environmental Protection Agency (USEPA 2005). In this approach, the carbon-based sorbent material is premixed with relatively inert materials such as clean sand or sediment and placed onto the contaminated sediment surface. Although this approach involves introducing materials in addition to the sorbent, it may have advantages at sites where a more spatially (vertically and laterally) uniform application of AC to the sediment surface is desired (because the AC can be mixed more thoroughly with the sand or sediment) or where more rapid control of HOC flux is desired.

Laboratory experiments and modeling studies (Murphy et al. 2006; Eek et al. 2008; Gidley et al. 2012), as well as field demonstrations (McDonough et al. 2007; Cornelissen et al. 2011, 2012) have confirmed the effectiveness of the blended cover application approach in reducing flux of mobile HOCs. At sites where additional isolation or erosion protection of underlying contaminated sediments may be needed, a related but separate option is to apply the sorbent as a layer within a conventional armored isolation cap. This paper, however, does not review either conventional or reactive isolation caps as defined by the USEPA (2005).

A full-scale example of blended AC application began in 2012 at Onondaga Lake, located in Syracuse, New York. The sediment cleanup remedy included placing bulk granular AC (GAC) blended with clean sand over approximately 110 hectares of lake sediments, along with related armored capping, dredging, and habitat restoration actions (NYSDEC and USEPA 2005; Parsons and Anchor QEA 2012). Full-scale implementation began following a successful field demonstration in fall 2011 and is currently scheduled to be completed in 2016.

Placing the blended GAC material in Onondaga Lake is being accomplished using a hydraulic spreading unit with advanced monitoring and control systems capable of placing approximately 100 cubic meters per hour of material in 6-meter-wide lanes (Figure 4). Granular AC amendment is mixed with sand and hydraulically transported and spread over sediment (average water depth of approximately 5 meters) through a diffuser barge. The GAC is presoaked for at least 8 hr prior to hydraulic mixing with the sand, to improve the settlement of the GAC through the water column. The spreader barge is equipped with an energy diffuser to distribute the blended materials evenly. The spreader barge incorporates electronic position tracking equipment and software so that the location of material placement can be tracked in real time. The spreader barge is also equipped with instruments for measuring the density of the slurry and the flow rates, which together provide the instantaneous production rate of the blended material being placed. Granular AC application rates are also tightly controlled and monitored using peristaltic metering pumps and a slurry density flow meter. The land-based slurry feed system is metered to the desired GAC dose.

Through the first 2 years of the 5-year construction project, the blended GAC material was placed in Onondaga Lake without any detectable losses to the water column. Verifying GAC placement was performed using both in situ catch pans located on the sediment surface prior to placement, as well as cores collected after placement. Results of these verifications demonstrated that the GAC was placed uniformly both horizontally and vertically within the sand layer applied to the lake (Supplemental Figure S7).

Figure 4. Hydraulic spreading application unit at Onondaga Lake, Syracuse, NY.

SITE EVALUATION AND DESIGN CONSIDERATIONS

The more than 25 field-scale demonstrations or full-scale projects performed through 2013 span a range of application methods and environmental conditions (including marine, brackish, and freshwater sites; tidal wetlands and mudflats; deep depths; steep slopes; under piers; and moving water [Table 1]). Collectively, these projects demonstrate the efficacy of in situ sediment treatment using sorptive, carbon-based amendments, particularly AC. As a result, in situ sediment treatment using AC is ready for full-scale application at a range of sites, subject to careful site-specific design analyses, generally as outlined in the next paragraphs.

To determine if site conditions are favorable for AC amendment, relatively simple bench testing of AC amendments can be performed by mechanically mixing AC into the sediments and performing straightforward porewater or bioaccumulation testing (e.g., Sun and Ghosh 2007). Short-term bench testing performed in this manner can rapidly identify sediment sites that are amenable to sediment treatment with AC and can be coupled with focused modeling or column studies to evaluate HOC behavior associated with groundwater flux. Bench testing can also be used to optimize AC materials (e.g., grain size or porosity) and dosing based on site-specific conditions. (Note that at most of the sites listed in Table 1, optimal AC doses were similar to the native organic carbon content of sediment.)

Although much has been learned to date, additional focused field-scale demonstrations may be particularly helpful to evaluate certain site-specific HOCs such as dioxins, furans, and methylmercury for which treatment effectiveness has been either variable or slow to develop (i.e., after the AC is mixed in) and in environments where sorptive carbon-based amendments have not yet been piloted (e.g., high-energy, erosion-prone locations). It is also important to note that at some sites, AC application may not provide additional protection compared to traditional sediment cleanup technologies. For example, mixing AC into a blended cover at Grenlandsfjords, Norway resulted in only marginal additional dioxin and furan flux reductions at 9 and 20 months compared with unamended clean sand or sediment cover materials, attributable in part to relatively slow sediment-to-AC transfer rates for large molecular volume dioxins and furans (Cornelissen et al. 2012; Eek and Schaanning 2012).

Based on a critical review of the results of the field-scale projects listed in Table 1, specific-site and sediment characteristics can reduce the effectiveness of AC application compared to other potential sediment cleanup technologies. These characteristics include (but are not likely limited to) relatively high native concentrations of black carbonaceous particles and slow sediment-to-AC transfer rates for relatively large molecular volume HOCs (Choi et al. 2014). Properly accounting for these and factors such as erosional forces and mixing or bioturbation in site-specific AC application design is necessary to ensure the effectiveness of the in situ remedial approach.

Experimental, modeling, and long-term monitoring lines of evidence from the case studies summarized in Table 1 have all confirmed that the effectiveness of AC applications increases over time at sites where there is not a significant flux from the underlying sediment to the surface. In many settings, full treatment effectiveness of AC amendments is achieved years after installation (e.g., Werner et al. 2006; Cho et al. 2012). The delay can be caused by (among other factors) the heterogeneity of AC distribution (even on a small scale), particularly at sites with relatively low bioturbation rates, as well as progressive mass transfer (Figure 5).

Site-specific evaluations of natural sediment deposition and bioturbation rates (as well as ongoing contaminant sources) and their effect on AC mixing and resultant restoration time frames are important design factors in developing appropriate site-specific in situ treatment strategies. Rates of natural sediment deposition and bioturbation-induced mixing of AC into the biologically active zone vary widely between sediment environments. For example, surface sediment bioturbation rates have been shown to vary more than 2 orders of magnitude between sediment environments, with relatively lower rates in wetlands and offshore sediments and relatively higher rates in productive estuaries and lakes (e.g., Officer and Lynch 1989; Wheatcroft and Martin 1996; Sandnes et al. 2000; Parsons and Anchor QEA 2012; Menzie et al. 2014). If relatively slow rates of natural deposition and mixing are anticipated, applying AC directly could be staggered over multiple applications to incorporate the amendment more evenly into the depositing sediments, albeit with potential cost implications.

As the USEPA (2005), NRC (2007), Bridges et al. (2010), ITRC (2014), and others have emphasized, the effectiveness of all sediment cleanup technologies depends significantly on sediment- and site-specific conditions. For example, resuspension and release of sediment contaminants occurs during environmental dredging, particularly at sites with debris and other difficult dredging conditions (Patmont et al. 2013). Optimizing risk management at contaminated sediment sites can often be informed by comparative evaluations of sediment cleanup technologies applied to site-specific conditions, considering quantitative estimates of risk reduction, risk of remedy, and remedy cost (e.g., Bridges et al. 2012). A hypothetical comparative risk reduction evaluation is presented in Figure 6 and highlights some of the short- and long-term tradeoffs that can occur between different sediment remediation technologies. Consistent with the example presented in Figure 6, at many sites, AC placement can achieve risk reductions similar to conventional capping but at a lower cost (see below), and may also provide better overall risk reduction than environmental dredging. Although Figure 6 presents a relatively common sediment remedial alternatives evaluation scenario in North America, it is important to note that site-specific conditions will result in varying risk reduction outcomes from alternative sediment remedies.

POTENTIAL NEGATIVE ECOLOGICAL IMPACTS

The acceptability of any sediment remediation option will depend on whether the benefits of the approach outweigh potential adverse environmental or ecological impacts, compared to other options. Because in situ treatment technologies involve adding a new material to sediments, in situ remedies have the potential to impact the native benthic community and vegetation, at least temporarily. A recent review by Janssen and Beckingham (2013) found that impacts to benthic organisms resulting from AC exposure were observed in one-fifth of 82 tests (primarily laboratory studies). Importantly, community effects have been observed more rarely in AC field pilot demonstrations compared to laboratory tests and often diminish within 1 or 2 years following placement (Cornelissen et al. 2011; Kupryianchyk et al. 2012), particularly in depositional environments where new (typically cleaner) sediment continues to deposit over time.

Although applying relatively higher AC doses or smaller AC particle sizes provide greater bioaccumulation reductions of HOCs, higher doses and smaller particle size may induce greater stress in some organisms (Beckingham et al. 2013). Negative impacts to benthic macroinvertebrates and aquatic plants resulting from adding AC, particularly at relatively high doses, may be attributable to nutrient reductions associated with AC amendment.

Although the available dose–dependent effects data for AC are not comprehensive, field trials and experimental studies suggest that potential negative ecological effects can be minimized by maintaining finer-grained AC doses below

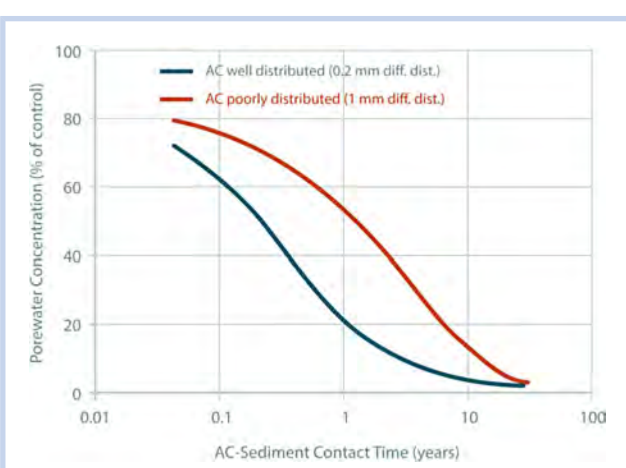


Figure 5. Model simulations of porewater PCB concentration reductions with different mixing scenarios (adapted from Cho et al. 2012).

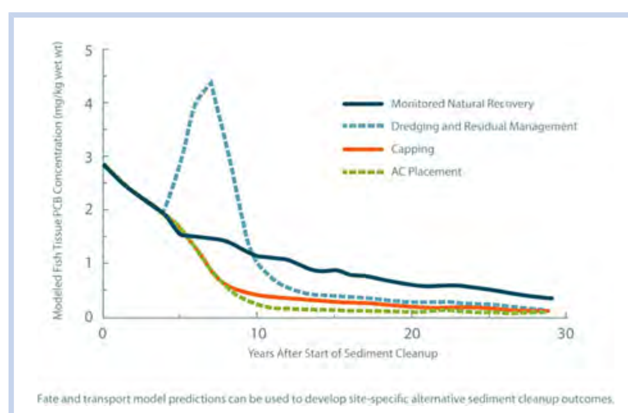


Figure 6. Hypothetical comparative net risk reduction of alternative sediment remedies. Example presented for illustrative purposes using the following fate and transport model input assumptions: average environmental dredge production rate of 400 m^3 per day and release of 3% of the PCB mass dredged (Patmont et al. 2013); average water flow through the cleanup area of 500 m^3 per second; implementation of effective upstream source controls; net sedimentation rate of 0.1 cm per year; and typical PCB mobility and bioaccumulation parameters.

Table 2. Summary of low- and high-range unit costs of AC application[a]

Component	Low-range Unit Cost	High-range Unit Cost
Activated Carbon[b]	$50,000/hectare	$100,000/hectare
Facilitating AC Placement Using Binder/Weighting Agents[c]	$0/hectare	$70,000/hectare
Facilitating AC Placement by Blending with Sediment or Sand[c]	$0/hectare	$100,000/hectare
Field Placement	$30,000/hectare	$200,000/hectare
Long-term Monitoring	$20,000/hectare	$100,000/hectare[d]
Total	$100,000/hectare	$500,000/hectare

[a]Estimated costs for a 4 percent AC dose (dry weight basis) over the top 10-cm sediment layer at a 5-hectare site.
[b]Powdered activated carbon (PAC) and/or granular activated carbon (GAC), depending on site-specific designs.
[c]To facilitate AC placement, binder or weighting agent amendments such as SediMite® or AquaGateTM, or clean sediment or sand (but typically not both) may be required in some applications depending on site-specific conditions and designs.
[d]High-end monitoring cost of $100,000 per hectare reflects prior pilot projects and likely overestimates costs for full-scale remedy implementation.

approximately 5% (dry wt basis; e.g., see discussion of the lower Grasse River AC demonstration). Similar to the net risk reduction comparisons summarized in Figure 6, the positive effects of reduced bioaccumulation of HOCs need to be balanced against potential negative short-term impacts. In addition, site-specific outcomes from in situ AC applications should be compared with outcomes resulting from other remediation approaches such as dredging and conventional capping, which are often greater than those resulting from in situ treatment.

RELATIVE SUSTAINABILITY OF DIFFERENT CARBON AMENDMENTS

Although amendments produced from different carbon source materials often exhibit similar effectiveness and negative ecological effects, different types of carbon amendments have different sustainability attributes. For example, life cycle analyses have demonstrated that AC produced from anthracite coal is less sustainable than AC produced from biomass feedstock (Sparrevik et al. 2011; e.g., agricultural residues), even though anthracite-derived AC may bind HOCs very effectively (Josefsson et al. 2012). One important positive effect of biomass AC related to sustainability is that its carbon is sequestered and removed from the global carbon cycle (Sparrevik et al. 2011). Even better sustainability outcomes can result from using non-activated pyrolyzed carbon, or “biochar” (Ahmad et al. 2014), because considerable amounts of energy are required for the activation process. However, the sorption capacity of biochars for many HOCs is more than an order of magnitude lower than AC (Gomez-Eyles et al. 2013).

COST

Based on a critical review of the field-scale projects listed in Table 1 for which adequate cost information was available, we summarized approximate low- and high-range unit costs for a full-scale AC application to a hypothetical 5-hectare sediment cleanup site. Cost summaries for the primary implementation components, not all of which may be needed at a particular site, are summarized in Table 2. Based on this summary, AC application is often likely to be less costly than either traditional dredging or capping approaches. Again, site-specific conditions can result in varying cost outcomes from alternative sediment remedies.

CONCLUSION

In situ sediment treatment using AC can rapidly address key exposures (e.g., bioaccumulation in invertebrates and fish), often becoming more effective over time due to progressive mass transfer. Due to its relatively large surface area, pore volume, and absorptive capacity, AC has a decades-long track record of effective use as a stable treatment medium in water, wastewater, and air. As such, AC is well suited for in situ sequestration and immobilization of HOCs in various sediment environments.

When designed correctly to address site-specific conditions, controlled (accurate and spatially uniform) placement of AC-bearing treatment materials has been demonstrated using a range of conventional construction equipment and delivery mechanisms and in a wide range of aquatic environments (Table 1), including wetlands. When contaminated sediments are present in unstable environments, traditional capping or dredging remedies might be the preferred option. Depending on sediment and site conditions, however, using AC can achieve short-term risk reduction similar to conventional capping and better overall risk reduction than environmental dredging, with lower costs and environmental impacts than traditional sediment cleanup technologies.

With a growing international emphasis on sustainability, in situ sediment treatment remedies offer an opportunity to realize significant environmental benefits, while avoiding the environmental impacts often associated with more invasive sediment cleanup technologies. Less invasive remediation strategies—such as treatment using in situ AC applications—are also typically far less disruptive to communities and stakeholders than dredging or conventional capping remedies. Important environmental, economic, and other sustainability issues can be associated with in situ sediment treatment, such as low-impact reduction of the bioavailable or mobile fractions of sediment contaminants through sequestration, improved recovery time frames, and reduced energy use and emissions (e.g., carbon; ITRC 2014).

Proper site-specific balancing of the potential benefits, negative ecological effects, and costs of in situ treatment relative to other sediment cleanup technologies is important to applying this approach successfully at full-scale. As discussed in USEPA (2005) and ITRC (2014), at most sites, a combination of sediment cleanup technologies applied to specific zones within the sediment cleanup site will result in a remedy that achieves long-term protection while minimizing short-term negative impacts and achieving greater cost effectiveness. It is evident from the extensive experimental studies and field-scale projects presented here that when applied correctly, in situ treatment of sediment HOCs using sorptive, AC-bearing materials has progressed from an innovative sediment remediation approach to a proven, reliable technology. Indeed, it is one that is ready for full-scale remedial application in a range of aquatic sites.

SUPPLEMENTAL DATA

Figure'S1. Simplified food chain model of in situ treatment.

Figure'S2. Pilot area and tine sled or tiller application units at lower Grasse River, NY.

Figure'S3. Dry broadcasting and slurry spray applications, Canal Creek, Aberdeen Proving Ground, MD.

Figure'S4. Vertical distribution of AC in wetland sediments at Canal Creek, Aberdeen Proving Ground, MD.

Figure'S5. SediMite® delivery at Mirror Lake, Dover, DE.

Figure'S6. Post-placement surface sediment AC concentrations at Mirror Lake, Dover, DE.

Figure'S7. Applied versus measured AC dose at Onondaga Lake, Syracuse, NY.

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ATTACHMENT B
ALTERNATIVE HYBRID REMEDIES (FROM EXPONENT, 2015)

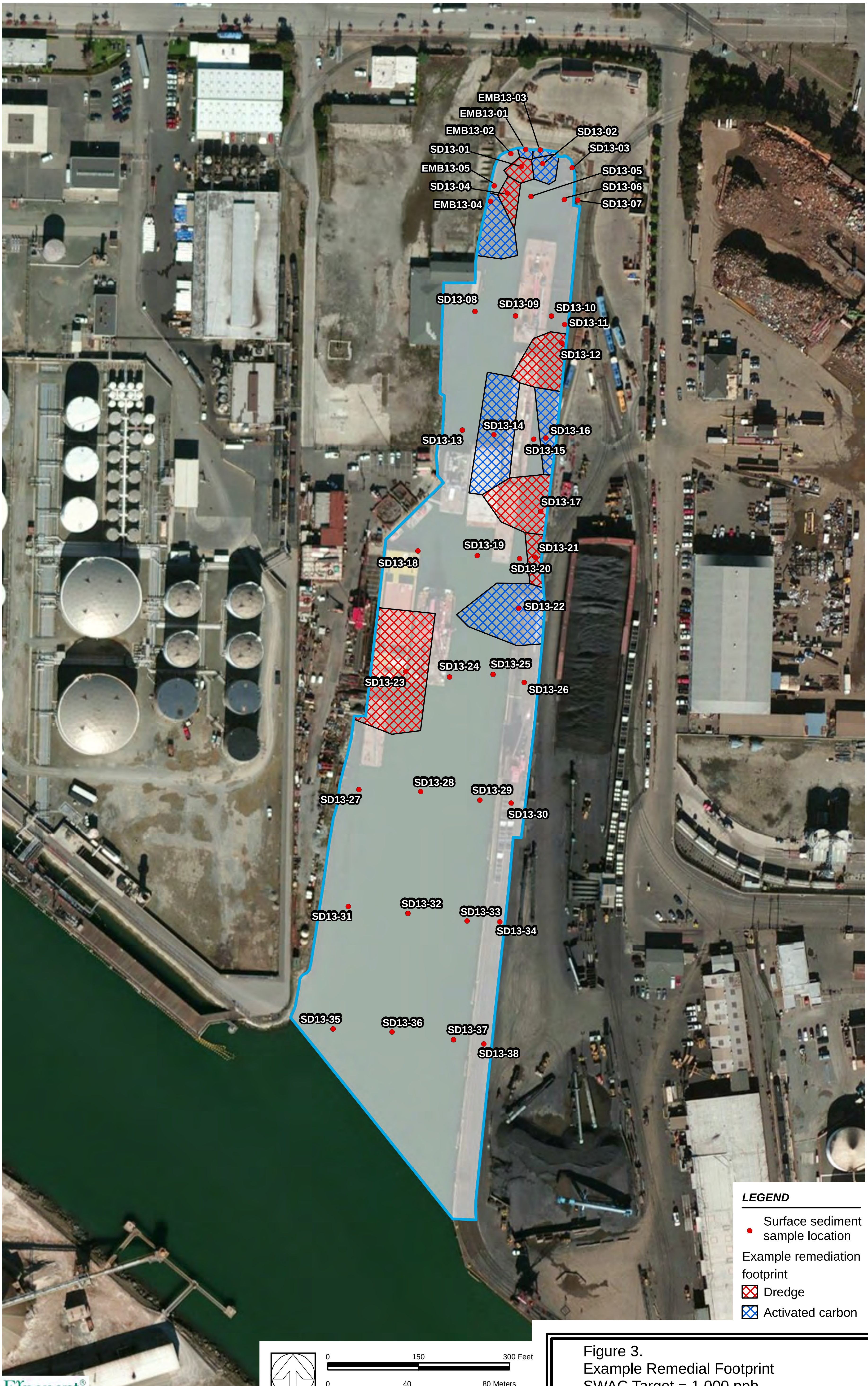

Figure 3.
Example Remedial Footprint
SWAC Target = 1,000 ppb
Carbon Efficiency = 95%

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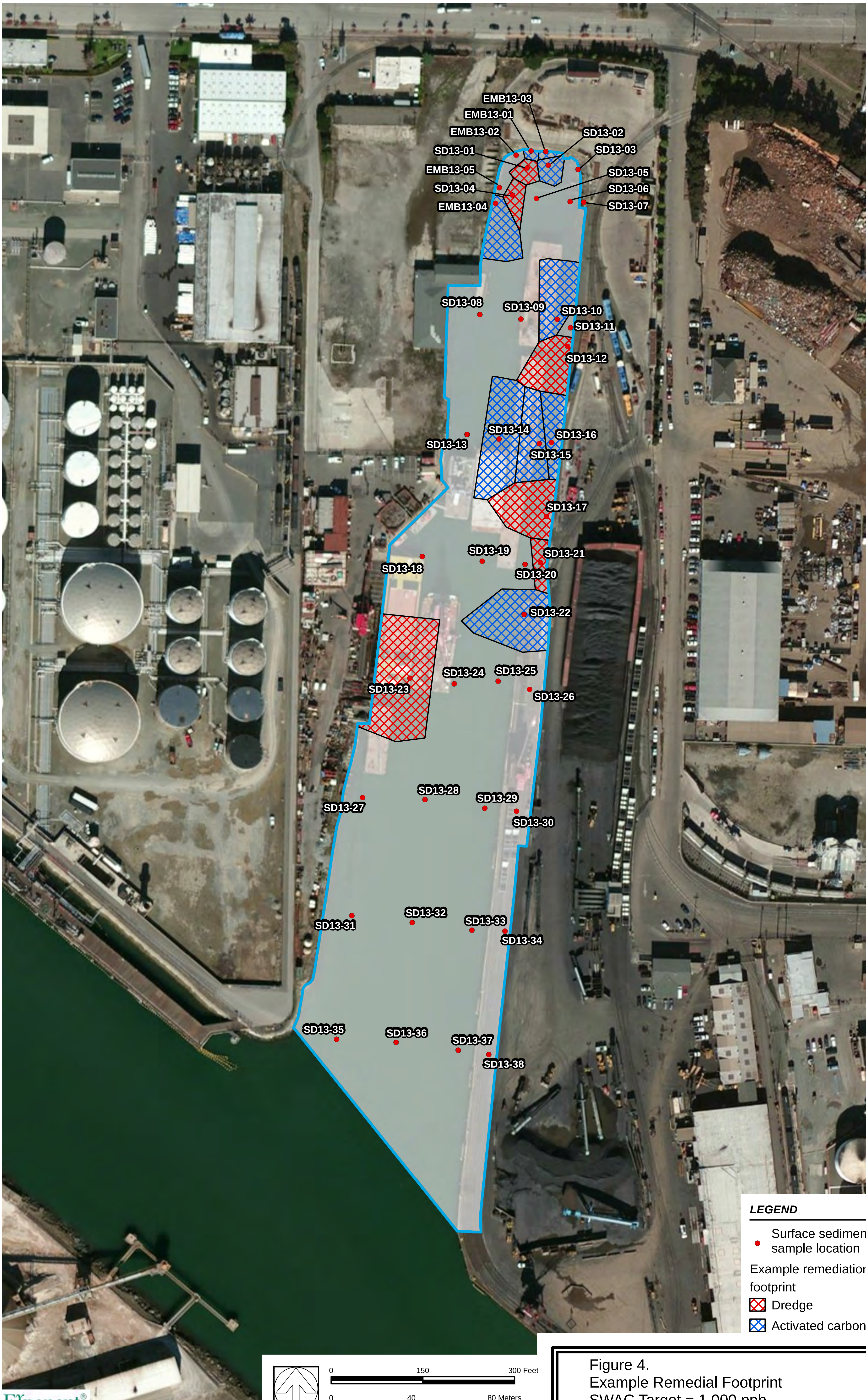

Figure 4.
Example Remedial Footprint
SWAC Target = 1,000 ppb
Carbon Efficiency = 80%

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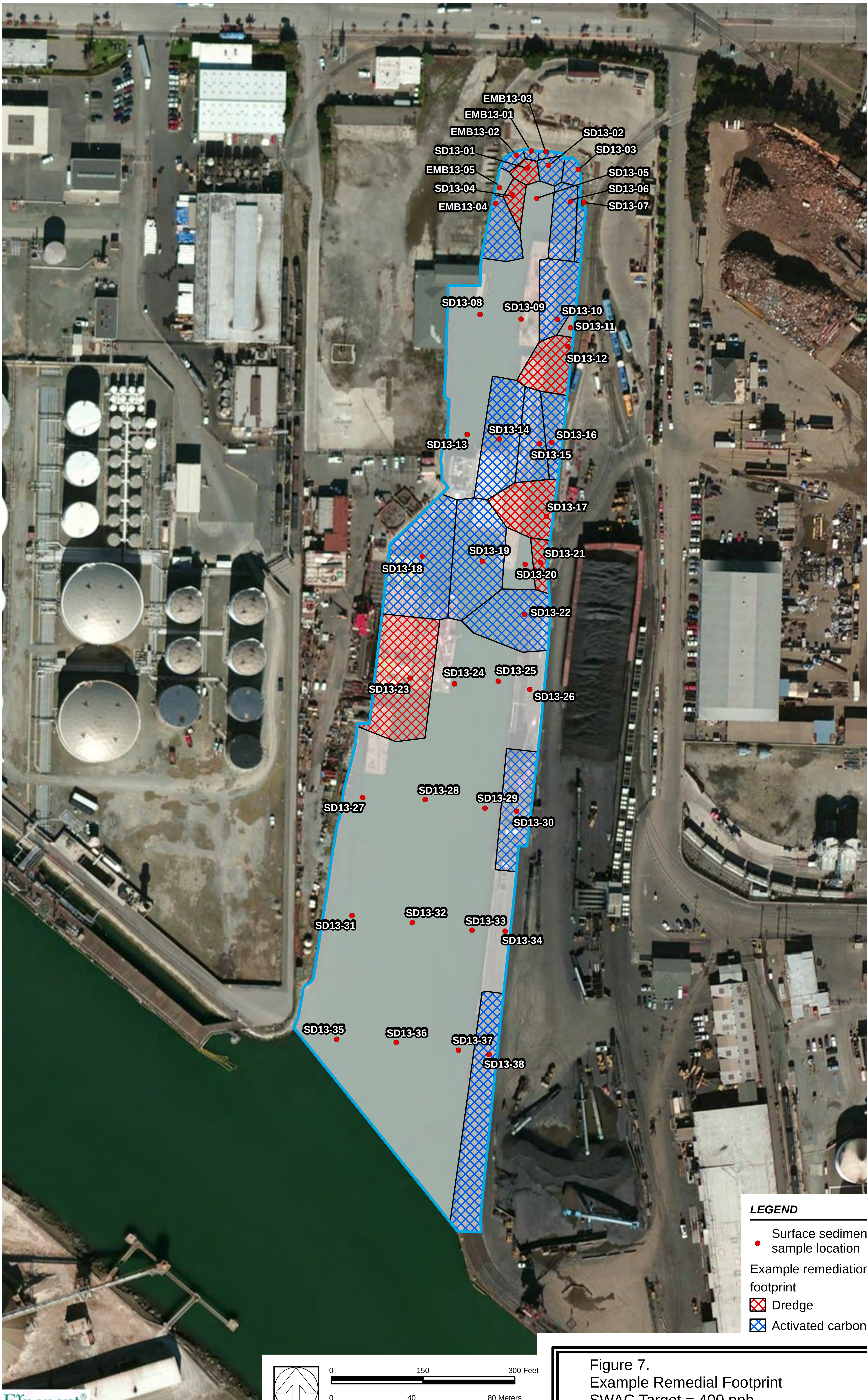

Figure 7.
Example Remedial Footprint
SWAC Target = 400 ppb
Carbon Efficiency = 95%

May 20, 2015 / P:\14\1406103.000_Parr_Richmond\GIS\projects\2013_locations - example remed footprints.mxd

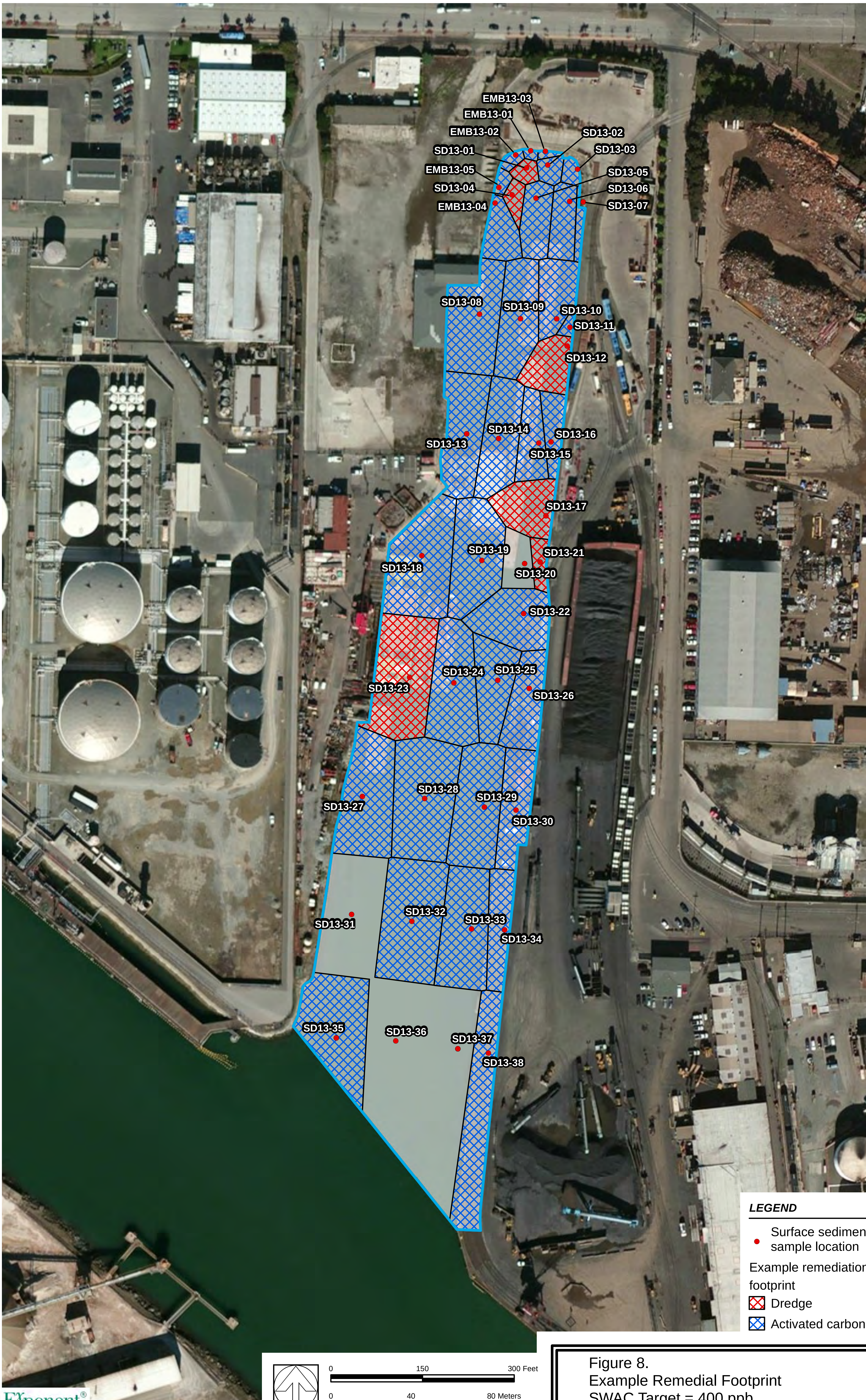

Figure 8.
Example Remedial Footprint
SWAC Target = 400 ppb
Carbon Efficiency = 80%

May 20, 2015 / P:\14\1406103.000_Parr_Richmond\GIS\projects\2013_locations - example remed footprints.mxd

Attachment C

Reassessment of Total DDT and Dieldrin Remediation Levels
United Heckathorn Superfund Site
Richmond, California

The U.S. Environmental Protection Agency (U.S. EPA) has conducted a reassessment of total DDT and dieldrin concentrations at the United Heckathorn Superfund Site. As part of this reassessment, two technical memorandums were prepared by CH2MHill that re-examine the fish consumption pathway of the human health risk assessment and the risks to ecological receptors. Technical comments on both reports are provided below.

Document:	**Technical Memorandum: Reassessment of Total DDT and Dieldrin Remediation Levels to Address the Fish Consumption Pathway at the United Heckathorn Superfund Site, Richmond, California**
Issued By:	**CH2M Hill**
Dated:	**February 10, 2010**

General Comments:

1. The assessment does not consider the central tendency exposure.

   The approach only provides an assessment of the reasonable maximum exposure (RME) scenario. The central tendency exposure (CTE) should also be provided. According to U.S. EPA guidance (1998)[1], the CTEs are intended to reflect central (more typical) estimates of exposure or dose. By providing both the RME and CTE, the magnitude of the exposure can be put into context and bounds can set for the risk estimates, although decisions are based on the RME (EPA, 1985)[2]. Providing the CTE exposure would help evaluate whether active or passive remediation actions should be considered. This concern is discussed further in the specific comments.

2. The Conceptual Site Model should be expanded and updated.

   The report should present a more detailed discussion of the current understanding of the source/pathway/receptor relationships at the Site. The Conceptual Site Model section references information in the 1994 Human Health Risk Assessment (HHRA) regarding fishing activities that may occur in the Study Area. More recent information should be used if possible to identify what populations are actually fishing in the Study Area and what type of fish are actually being caught and consumed.

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[1] Environmental Protection Agency (EPA). 1989. Risk Assessment Guidance for Superfund, Volume 1: Human Health Evaluation. Manual (Part A, Baseline Risk Assessment). USEPA/540/1-89/002. Office of Emergency and Remedial Response, Washington D.C.

[2] Environmental Protection Agency (EPA). 1985. National Oil and Hazardous Substances Pollution Contingency Plan. Final Rule. 50 Federal Register 47912 (November 20, 1985).

3. The fish tissue data used to assess the fish consumption pathway may be inappropriate.

   The shiner surfperch is an important bait fish (CA Fish and Game) in San Francisco Bay. They are used as bait to catch striped bass and California halibut, but have a low value as a sport or human food fish. The small sample size combined with the non-edible size fish and whole body analysis does not provide a reasonable estimate for human exposure. This is more appropriate for an ecological assessment and may be better provided as a portion of a food web to higher trophic level consumers. These concerns are discussed further in the specific comments.

Specific Comments:

1. Section 3.1 - 1994 Human Health Risk Assessment.

   As part of the summary of the 1994 HHRA, fishing in Lauritzen Channel is described as being restricted, fenced by industrial facilities and warning signs posted in several languages. In 1994 these institutional controls were deemed inadequate, and since then access restrictions have been established to control fishing activities. This is in addition to the fact that the channel has a high level of barge and tug boat activity would limit the fishing potential in the channel.

2. Section 4.3 - Potentially Exposed Populations.

   The authors report that of all the fish, the white croaker and perch are the most popular in the Parr Canal area. However, this is not supported by any of the references cited. The *San Francisco Bay Seafood Consumption Study* (CDHS, 2000)[3] and *A Seafood Consumption Survey of the Laotian Community of West Contra Costa County, California* (APEN, 1998)[4] report that striped bass and sturgeon are caught and consumed more than white croaker or perch. In fact, no white croakers were caught during the 2008 fish collection effort.

   In addition, the authors cite information from the APEN report regarding Laotian practices of food preparation such as the making of sauces and paste or eating whole fish that could result in higher exposures. However, it is not discussed whether the Laotians may be fishing within the Study Area and in fact using these fish for these purposes. The APEN study would indicate otherwise. The study found that the most popular fishing location was the San Pablo Dam with the next most popular places mentioned as Lake Sonoma and Point Pinole. Approximately 22% of survey respondents indicate they fish in the San Francisco Bay with a variety of locations mentioned. The closest location to the Study Area, Marina Bay, was mentioned once. The survey also reported that, while 25.7% of the survey respondents (of those who eat sauces and pastes) make sauces at home the majority (95.6%) buy their sauces from stores or restaurants.

[3] California Department of Health Services (CDHS). 2000. San Francisco Bay Seafood Consumption Report. Environmental Health Investigators Branch.

[4] Asian Pacific Environmental Network (APEN). 1998. A Seafood Consumption Survey of the Laotian Community of West Contra Costa County, California. March.

3. Section 5.0 - 2008 Fish Collection & Analysis.

   The sample size for fish is relatively low for this type of assessment. In the companion document to this, *Reassessment of Risk to Ecological Receptors*, U.S. EPA and its contractor supplemented these data with data available from the California Sediment Quality Objective Database. This data source should also be considered in this assessment.

4. Section 5.0 - 2008 Fish Collection & Analysis.

   The data collected and used in this assessment are not reflective of fish likely to be consumed by people. With the exception of the anchovy, the majority of fish were too small to fillet and therefore analyzed as whole fish or added to a composite of whole fish. DDT and dieldrin are highly hydrophobic and partition into the lipids of biological tissues. As a result, organs such as the liver and intestine will preferentially accumulate these compounds. In most cases, these organs are not consumed. The San Francisco Bay Seafood Consumption Study (CDHS, 2000) reported only 1% of the consumers reported eating "guts" and only 25% consumed the skin. In the APEN study over half (53.4%) report they never eat the organs. Including whole body measurements over estimates human exposures to DDT and dieldrin.

5. Section 6.1.1 – Fish Consumption Rates.

   The fish consumption rate of 85.1 grams per day was used in the risk assessment. This value is based on the 95th percentile of the responses from all people surveyed in the APEN study (APEN 1998). A median value of 9.1 grams per day, which is considered a more accurate measure of central tendency or average exposure, is also reported in the APEN study. At a minimum, this value should be used in an estimate of central tendency risks in the risk assessment. The APEN values along with other values from relevant studies should be discussed in the risk assessment to present the range of consumption rates that have been observed. There is mention of a high-end estimate from OEHHA's report (OEHHA, 2008) in the uncertainty section. However, the full range of possible fish consumption rates should be discussed not just the high end.

6. Section 6.1.2 - Fraction of Total Diet Coming from Study Area.

   The portion of fish consumed from the Study Area is over estimated. Although the approach accounts for that portion of the diet obtained from store bought sources, approximately 50%, it fails to account for the percentage of fishing taking place in the Bay and the method of catching fish. Less than 25% of Laotian community fish in San Francisco Bay; San Pablo Dam and Lake Sonoma are the most popular places to fish (APEN, 1998). The closest location to the Study Area mentioned in the APEN study (Marina Bay) was mentioned only once. In addition, depending on the ethnic group evaluated, 35% - 65% catch their fish from piers, a method which is highly constrained by industrial activities in the Study Area (CDHS, 2000).

7. Section 6.1.3 - Exposure Point Concentrations.

   The exposure point concentration (EPC) fails to account for the loss of DDT and dieldrin from the cooking process. In the uncertainty section, the authors recognize that CalEPA (2008)[5] assumes 50% of the pesticide concentration is lost during cooking. In the Laotian community it is common tradition to prepare sauces and pastes from shrimp, crab, or fish (*e.g.*, fish pudding or lap). These condiments can be made from cooked or raw seafood. However, 96% of the community surveyed (APEN, 1998)[5] reported that they buy their fish pudding or lap. This would indicate that cooking losses should be considered in developing EPCs. CalEPA's recommendation is further supported by guidance established by U.S. EPA (2000) [6] and summarized in the following table.

Summary of Contaminant Percent Loss[a] from Fish Due to Cooking

Contaminant	Skin-Off			Skin-On		
	Minimum	Average	Maximum	Minimum	Average	Maximum
DDD	4	30	88	10	37	54
DDE	7	30	75	7	39	59
DDT	0	38	141	4	33	60
Dieldrin	4	29	3	3	36	93

a. Percent losses are derived by combining all cooking method

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[5] California Environmental Protection Agency (CalEPA). Office of Environmental Health Hazards Assessment. 2008. Fish Contaminant Goals and Advisory Tissue Levels for Contaminants in Sport Fish.

[6] Environmental Protection Agency (EPA). 2000. Guidance for Assessing Chemical Contaminant Data for Use In Fish Advisories. Volume 2: Risk Assessment and Fish Consumption Limits - Third Edition. Appendix C. Dose Modifications Due to Food Preparation and Cooking. November. EPA 823-B-00-008.

8. Section 6.3.4 – Risk Characterization Results.

   The results for each water body are presented in this section. However, there is no discussion on the likelihood of exposures actually occurring. For example, the highest reported risks are associated with the Lauritzen Channel where there is the lowest exposure potential due to the limitations in access.

9. Section 8.1 - Derivation of Sediment to Biota Regression Relationships.

   The development of a regression relationship to predict bioaccumulation would benefit from a more robust data set. Data available from SFEI, used in this assessment to examine background concentrations should also be incorporated into the regression analysis. In using only data from the Study Area the results are biased; there are no data points for the lower concentration range. As a result, the resulting model may overestimate the potential bioaccumulation potential.

10. Section 8.1 - Derivation of Sediment to Biota Regression Relationships.

    The values in Tables 10 and 11 are based on a regression model for surfperch using wet-weight fish tissue concentrations and dry-weight sediment concentrations. It is not explained why this model is preferred over the approach that accounts for sediment organic carbon or fish tissue lipid content (as described in the text and presented in the table below). Sediment and fish tissue data should be normalized to sediment organic carbon and tissue lipid. This approach was used by staff from CH2MHill and SFEI in a recent publication (Melwani et al., 2009)[7] examining bioaccumulation, including surfperch in San Francisco Bay.

Type of RBC	Concentration in Sediment (ug/kg dw)	
	DDT	Dieldrin
Sediment-to-biota regression-based RBCs	450	13
Lipid-TOC BSAF RBC	575	16

11. Section 8.1 - Derivation of Sediment to Biota Regression Relationships.

    Please explain the two sets of RBC_{fish} values identified for each compound and each target risk presented in Table 7. Which set of values were used to determine the $RBC_{sediment}$ presented in Table 10? This is unclear.

[7] Melwani, Aroon, Ben Greenfield, and Earl Byron. 2009. Empirical Estimation of Biota Exposure Range for Calculation of Bioaccumulation Parameters. Integr Environ Assess Manag. 5:138-149.

12. Section 8.1 - Derivation of Sediment to Biota Regression Relationships.

    Additional rationale should be provided as to the selection of the shiner surfperch for the regression model. The selection of the appropriate species should be based on the current conceptual site model and what people are actually consuming if the end goal is protection of human health.

Document: **Draft Reassessment of Remediation Levels to Address Risks to Ecological Receptors at the United Heckathorn Superfund Site, Richmond, California.**
Issued By: **CH2M Hill**
Dated: **February 9, 2010**

General Comments:

1. The reassessment approach uses a "shot gun" method at modeling bioaccumulation to ecological receptors. Three different models are used, two different data sets, and different exposure pathways are considered. From these results, the lowest values are identified. This is not an acceptable approach.

   The Trophic Trace model uses a different data set and incorporates different exposure pathways, including pathways identified in the conceptual site model as possible and of low significance (*i.e.*, surface water). Each model has its own strengths and weaknesses. No attempt is made to determine which method is applicable to this assessment. In an attempt to validate the Trophic Trace the authors compared the predicted concentrations to measured concentrations and erroneously concluded that because there was an overlap in concentration ranges that the model was a reasonable predictor. The modeled concentration for DDT in water ranged up to 2,202 ng/L while the maximum measured concentration was 63.1 ng/L. Similar comparisons of tissue concentrations yielded the same observations. For example, the mean predicted tissue concentration of DDT in Bay Shrimp 1,670 μg/kg ww while the mean measured concentration was 32 μg/kg ww and the maximum concentration was 46.7 μg/kg ww. Given the site-specific data available for this assessment, the Trophic Trace model is not appropriate for this assessment.

   Choosing an appropriate model should be based on applying an understanding of the physico-chemical relationships between hydrophobic compounds in sediment and tissues, uptake mechanisms, and available data. Using organic carbon normalized concentrations is consistent with U.S. EPA's guidance. As such, the regression model using lipid normalized and total organic carbon normalized sediment data should be considered the technically appropriate method.

2. The exposure factors used to estimate site use are appropriate for a general screening-level ecological risk assessment, but not appropriate for a focused feasibility or reassessment in which decisions regarding remedial action are being considered. In each iterative step through the risk assessment process, more site-specific data and refinements should take place to reduce the level of uncertainty and improve the confidence in the resulting management decisions. This reassessment does not accomplish this goal. These concerns are addressed further in the specific comments that follow.

3. The authors of the technical memorandum (TM) state that the purpose of the TM is to:

> "…update the existing ecological risks information associated with the United Heckathorn site. The other purpose of the TM was to use existing and recently collected data to identify concentrations of total dichloro-diphenyl-trichloroethane (DDT) and dieldrin in sediment and biota that are associated with no or minimal effects to selected ecological receptors."

However, later in the same paragraph they state that given the elevated concentrations in sediment and biota in recent samples, the presence of ecological risk was assumed and an updated ecological risk assessment was not performed. Considering that the companion TM on the fish consumption pathway reassessed the potential for unacceptable human health risks using current data, it seems reasonable that the ecological risk reassessment should follow the same methodology.

The ecological risk reassessment should use reasonable exposure parameters and models consistent with a baseline risk assessment and not those used in conservative screening-level risk assessments. This reevaluation can then be used to refine the risk assessment and focus the Focused Feasibility Study on those geographical areas and receptors that are at risk.

This should also include a more detailed discussion of the current understanding of the source/pathway/receptor relationships at the Site. The development of this conceptual site model would be one outcome of an ecological risk assessment and would provide context for the exposure pathways and receptors used to develop ecologically-based safe sediment concentrations. Additionally, an accurate assessment of the conceptual site model complete with an understanding of the actual sources, pathways and relationships of the receptors for various compartments at the site is important for future remedial design work. Relying on the old concept of removing bulk sediment concentrations of contaminants, without altering the relationships of contaminants to organic carbon is likely to result in the same position – no improvement in the tissues of organisms after dredging, removal and capping of residuals that was performed over a decade ago.

Specific Comments:

1. Section 7.0 - Available Data Used for this TM.

This section ends with a statement that the fish collected within the Lauritzen Channel had significantly higher concentrations for DDT and dieldrin. However, the range of concentrations from each area overlaps. The information needed to support this statement was not provided. The authors should provide mean lipid normalized tissue concentrations and the standard deviation for the fish species that are being compared.

2. Section 8.0 - Derivation of Sediment-to-Biota Bioaccumulation Models.

   As discussed in this section, it is appropriate to look at total organic carbon (TOC) normalized sediment values and lipid normalized tissue values when assessing bioaccumulation of hydrophobic compounds. It is unclear why a similar approach was not used in the companion Fish Consumption document prepared by CH2MHill.

3. Section 8.0 - Derivation of Sediment-to-Biota Bioaccumulation Models.

   The areas and the associated sediment chemistry used in this model need to be included in a summary table. The numbers cannot be reproduced without these values. Additionally, figures showing the location of the trawls in relation to the sediment sampling locations used to estimate sediment concentrations are needed. This would allow reviewers to independently confirm the representativeness of the sediment data used in developing biota sediment accumulation factors (BSAFs). Considering the strong sediment concentration gradient within the Lauritzen Channel north to south, lumping all that data together and taking the median concentration as being representative of sediment within the channel is introducing significant uncertainty into the model.

4. Section 8.0 - Derivation of Sediment-to-Biota Bioaccumulation Models.

   Please confirm that the additional data obtained from the California Sediment Quality Database was used in the development the regression models.

5. Section 8.0 - Derivation of Sediment-to-Biota Bioaccumulation Models.

   In most cases, only a few sediment concentrations were used to develop BSAFs or the regression bioaccumulation models. This is because sediment data from within a geographic area was lumped together and the median concentration was used to represent the entire area (see Comment #3). All biota sampled within that area was then assigned the median sediment concentration in the models. As shown in Tables 7 and 8, the high variability in BSAFs (in a few instances three orders of magnitude between the minimum and maximum BSAF calculated for a specific species) may be due in part to the uncertainty in sediment concentrations. Additionally, when the regression models are used, significant uncertainty is introduced for those cases where you may have a reasonable tissue sample size (greater than 10), but the sediment sample size may only be two or three data points. Thus a regression is being developed based on a very few data points, introducing additional uncertainty.

6. Section 9.1 - Risk-Based Sediment Contaminant Concentrations for Fish and Invertebrates.

   The acceptance of a regression model with a correlation coefficient (r^2) less than 0.50 is not technically justified. An r^2 of 0.20 would indicate that there is a high degree of variability that is not captured by the model. Given the use of these models is to determine risk-based cleanup levels, a much higher degree of correlation coefficient should be expected, such as 0.75.

7. Section 9.1 - Risk-Based Sediment Contaminant Concentrations for Fish and Invertebrates.

   The tissue effect levels identified in Table 11 are not appropriate for San Francisco Bay or the receptors of concern that were identified. The dieldrin fish effect threshold is based on a sensitive freshwater species, the rainbow trout. The DDT threshold for fish is not even identified and the effects are vaguely described as "primarily mortality" with a "minimal" magnitude of effect from unpublished data. Given the magnitude of studies conducted on DDT and dieldrin over the last four decades, it seems reasonable that a more appropriate data set for estuarine species would be available and that it may be possible to look at thresholds for both benthic and pelagic fish species. These thresholds are being used to determine possible remedial actions; therefore, a more robust data set is warranted.

8. Section 9.2 - Risk-Based Sediment Contaminant Concentrations for Birds and Mammals.

Since a formal reevaluation of ecological risks was not presented in this TM, it is unclear how the three bird species (surf scoter, double-crested cormorant and the Forster's tern) and the one mammal (harbor seal) were chosen as receptors of interest to develop risk-based sediment concentrations. Generally, risk-based sediment concentrations are developed for species that are linked in some way to the assessment endpoints identified in the risk assessment and in the development of remedial action objectives. This document presents no discussion as to why these species are relevant to the reevaluation except that they are, "resident species in San Francisco Bay, may forage in and around the United Heckathorn site, and therefore may be exposed to contaminants…".

9. Section 9.2 - Risk-Based Sediment Contaminant Concentrations for Birds and Mammals.

   Little documentation is presented in the TM on how risk-based concentrations were calculated. As such, it was impossible to recreate the approach from the information provided. Some examples are provided below:

   a) Life history parameters for the species of interest were presented in Tables 14 through 16 without sufficient justification. Similar species have been used at other sites in San Francisco Bay and different parameters have been selected. For example, Hatch and Weseloh (1999)[8] report a range of ingestion rates from 208 to 537 g/day wet weight with an average of about 320 g/day wet weight for adult birds. This FIR is significantly lower than the one used in the TM and would result in a higher risk-based sediment concentration.

   b) It is not possible with the information presented to confirm that the wet weight (ww) and dry weight (dw) units conversions were conducted correctly. For example, it appears that the LOAEL TRV for birds from the US EPA EcoSSLs is in mg dw/kg bw-d. However, it seems that the FIR and prey concentrations are in wet weight. If in fact these units are different, another conversion may need to be conducted.

10. Derivation of Risk-based Concentrations in Sediment Based on the Trophic Trace Mechanistic Model

    Site-specific BSAFs were used in the trophic trace model. A number of uncertainties have been identified with the development of these BSAFs (see comment 5). These uncertainties would also impact the output of the trophic trace model.

11. Derivation of Risk-based Concentrations in Sediment Based on the Trophic Trace Mechanistic Model

    The trophic trace model consistently overestimates the tissue concentrations modeled; up to three orders of magnitude are observed between the site-specific measured tissue concentrations and the modeled concentrations (Table 19). There is no discussion in the

[8] Hatch, J.J., and D.V. Weseloh. 1999. "Double-Crested Cormorant". In: *The Birds of North America*. Eds. A. Poole and F. Gill. No. 441. Pp 1-35.

TM explaining the poor relationship between measured and modeled concentrations. Considering the poor performance of the model as an estimator of tissue concentrations, the model is not appropriate to use in the development of risk-based cleanup goals.

12. Uncertainties.

This evaluation assumes that the all the receptors feed exclusively in the Study Area. Although this is expected and often appropriate in a screening-level ecological risk assessment, it is not appropriate or warranted as part of a Focused Feasibility Study or determining the need for remedial actions. The Study Area that is roughly 22 acres and the Lauritzen Channel is approximately 5 acres. Most of the receptors forage in areas larger than the Study Area, are limited by the poor quality and availability of suitable habitat in this highly industrialized area, and many are migratory and are not the region for half of the year. A site use factor (SUF) of 1.0 cannot be scientifically supported. Reasonable SUFs should be developed and applied to this reassessment.

- The surf scoter is not a yearlong resident and according to California Department of Fish and Game, they migrate to winter breeding grounds in northern Canada and Alaska. In addition, the surf scoter does not nest in California. While on their wintering grounds, scoter forage over areas larger than the Study Area. A two-year radiotelemetry study conducted in the Commencement Bay Area of Puget Sound found that wintering birds stayed within 9 to 11 kilometers of their capture location (Mahaffy et al., 1995[9]). Thus the application of a SUF of 1.0 is not technically appropriate.

- Cormorant is also not a year round resident of the Bay. According to California Department of Fish and Game, cormorants spend the summer in the mountains and northeast plateau. In addition, their home range can be up to 5-10 mile radius of their nest which requires an undisturbed area beside the water. In San Francisco Bay, double-crested cormorant breeding at the Richmond-San Rafael Bridge foraged within 5 kilometers of the bridge (Stenzel et al., 1995). This is a significantly larger area than the Study Area. The application of a SUF of 1.0 is not supported by the cormorant's life history.

- Forster's terns migrate seasonally, spending 7 months of the year in south California to South America. According to California Department of Fish and Game, these birds nest on isolated levees and islands scraping a small depression in the soil for a nest. The Study Area does not provide the required habitat, as such, a SUF of 1.0 is not appropriate.

- The Harbor Seal, although a yearlong resident, its foraging range is dependent on suitable haul areas. Harbor seals have been known to travel moderate to long

[9] Mahaffy, M.S., C.R. Bennett, and W. Parsons. 1995. "Puget Sound Waterbirds as Contaminant Monitors." *Puget Sound Research '95, Proceedings*. Puget Sound Water Quality Authority. January 12-14.

distances based on the availability of prey. Within the channelized Study Area, suitable haul out areas are unlikely. It is not reasonable to apply a SUF equal to 1.0 for the Harbor Seal.

13. Summary and Conclusions.

This section is confusing. It is unclear why there is a discussion on why the surfperch is considered the best species on which to base the regression equation, when the definition of the "best species" is dependent on which species you are interested in protecting. If the surf perch is the best species then why are risk-based sediment concentrations calculated for a variety of receptors using three different approaches (Table 21)? While the last paragraph of the text indicates that no specific remedial target is recommended in the TM, Table 21 draws attention to the most conservative ecological risk-based sediment concentrations through shading.

14. 12.0 - Summary and Conclusions.

The authors indicate that risk-based sediment concentrations were developed for 18 fish and wildlife receptors; however, the basis for these 18 risk-based sediment values are three different models and only four toxicity threshold values, two for DDT and two for dieldrin. As discussed previously, this "shot gun" approach is not appropriate for a Focused Feasibility Study.

Attachment D

Integrated Environmental Assessment and Management — Volume 9999, Number 9999—pp. 1–13
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Critical Review

In Situ Sediment Treatment Using Activated Carbon: A Demonstrated Sediment Cleanup Technology

Clayton R. Patmont,†‡ Upal Ghosh,§ Paul LaRosa,|| Charles A. Menzie,# Richard G. Luthy,†† Marc S. Greenberg,‡‡ Gerard Cornelissen,§§|||| Espen Eek,§§ John Collins,## John Hull,## Tore Hjartland,††† Edward Glaza,‡‡‡ John Bleiler,§§§ and James Quadrini||||||*

†Anchor QEA, Seattle, Washington, USA
‡Present address: Anchor QEA, 720 Olive Way, Suite 1900, Seattle, WA 98101
§University of Maryland Baltimore County, Baltimore, Maryland, USA
||Anchor QEA, Amesbury, Massachusetts, USA
#Exponent, Alexandria, Virginia, USA
††Stanford University, Stanford, California, USA
‡‡US Environmental Protection Agency, Edison, New Jersey, USA
§§Norwegian Geotechnical Institute, Oslo, Norway
||||Norwegian University of Life Sciences, Ås, Norway
##AquaBlok, Toledo, Ohio, USA
†††BioBlok Solutions, Sandefjord, Norway
‡‡‡Parsons, Syracuse, New York, USA
§§§AECOM, Chelmsford, Massachusetts, USA
||||||Anchor QEA, Woodcliff Lake, New Jersey, USA

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ABSTRACT

This paper reviews general approaches for applying activated carbon (AC) amendments as an in situ sediment treatment remedy. In situ sediment treatment involves targeted placement of amendments using installation options that fall into two general approaches: 1) directly applying a thin layer of amendments (which potentially incorporates weighting or binding materials) to surface sediment, with or without initial mixing; and 2) incorporating amendments into a premixed, blended cover material of clean sand or sediment, which is also applied to the sediment surface. Over the past decade, pilot- or full-scale field sediment treatment projects using AC—globally recognized as one of the most effective sorbents for organic contaminants—were completed or were underway at more than 25 field sites in the United States, Norway, and the Netherlands. Collectively, these field projects (along with numerous laboratory experiments) have demonstrated the efficacy of AC for in situ treatment in a range of contaminated sediment conditions. Results from experimental studies and field applications indicate that in situ sequestration and immobilization treatment of hydrophobic organic compounds using either installation approach can reduce porewater concentrations and biouptake significantly, often becoming more effective over time due to progressive mass transfer. Certain conditions, such as use in unstable sediment environments, should be taken into account to maximize AC effectiveness over long time periods. In situ treatment is generally less disruptive and less expensive than traditional sediment cleanup technologies such as dredging or isolation capping. Proper site-specific balancing of the potential benefits, risks, ecological effects, and costs of in situ treatment technologies (in this case, AC) relative to other sediment cleanup technologies is important to successful full-scale field application. Extensive experimental studies and field trials have shown that when applied correctly, in situ treatment via contaminant sequestration and immobilization using a sorbent material such as AC has progressed from an innovative sediment remediation approach to a proven, reliable technology. *Integr Environ Assess Manag* 2015; 9999:XX–XX. © 2014 The Authors. Published 2014 SETAC.

Keywords: Activated carbon Sediment In situ treatment Bioavailability Remediation

All Supplemental Data may be found in the online version of this article.
* To whom correspondence may be addressed: cpatmont@anchorqea.com
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DOI: 10.1002/ieam.1589

KEY POINTS

- More than 25 field-scale pilot or full-scale sediment treatment projects performed over the past decade, along with numerous laboratory experiments, have proven the efficacy of in situ sediment treatment using AC to reduce the bioavailability of several hydrophobic organic compounds.
- Controlled placement of AC (accurate and spatially uniform) has been demonstrated using a variety of conventional construction equipment and delivery techniques and in a range of aquatic environments including wetlands.
- In situ sediment treatment using AC has progressed from an innovative remediation approach to a proven, reliable

technology that is ready for full-scale application at a range of sites.

INTRODUCTION

Sediments accumulated on the bottom of a waterbody are recognized as sinks for toxic substances and bioaccumulative chemicals and can be long-term reservoirs for chemicals that can be transferred via the food chain to invertebrates and fish (USEPA 2005). Establishing effective methods to reduce the ecological and human health risks contaminated sediment poses has been a regulatory priority in North America, Europe, and elsewhere since the 1970 s. Indeed, demonstrating risk reduction that is convincing to all stakeholders using traditional dredging and isolation capping approaches has been challenging (NRC 2007; Bridges et al. 2010). Although traditional approaches will continue to be an integral part of sediment cleanup remedies (e.g., when contaminated sediments are present in unstable environments), new remediation approaches are needed to either supplement or provide alternatives to existing methods.

In situ sediment treatment via contaminant sequestration and immobilization generally involves applying treatment amendments onto or into surface sediments (Luthy and Ghosh 2006; Supplemental Figure S1). This paper reviews the considerable advances in engineering approaches used to apply activated carbon (AC)-based treatment amendments in situ; summarizes field-scale demonstration pilots and full-scale applications performed through 2013; and describes lessons learned on the most promising application options. This paper also discusses the need for a balanced consideration of the potential benefits, ecological effects, and costs of in situ treatment using AC relative to other sediment cleanup technologies. The results of this work aim to identify a common set of features from engineering, chemistry, and ecology that could help guide and advance the use of in AC-based in situ sediment treatment in future sediment remediation projects.

TREATMENT AMENDMENTS AND MECHANISMS

Beginning in the early 2000s, encouraging results from laboratory tests and carefully controlled, small-scale field studies generated considerable interest in remediating, or managing, contaminated sediments in situ. Mechanisms to do so mainly suggested sorptive treatment amendments such as AC, organoclay, apatite, biochar, coke, zeolites, and zero valent iron (USEPA 2013a). Three of these amendments—AC, organoclay, and apatite—have been identified as particularly promising sorptive amendments for in situ sediment remediation (USEPA 2013b). Of these, AC has been used more widely in laboratory experiments and field-scale applications to control dissolved hydrophobic organic compounds (HOCs). This is largely because AC has been used successfully for decades as a stable treatment medium for water, wastewater, and air, and because early testing of sediment treatment with AC showed positive results.

Laboratory testing and field-scale applications of AC have demonstrated its effectiveness in reducing HOC bioavailability. Both natural and anthropogenic black carbonaceous particles in sediments, including soot, coal, and charcoal strongly bind HOCs, and the presence of these particles in sediments has been demonstrated to reduce biouptake and exposure substantially (Gustafsson et al. 1997; Cornelissen et al. 2005). Using engineered black carbons such as AC augments the native sequestration capacity of sediments, resulting in reduced in situ bioavailability of HOCs. When AC is applied at optimal, site-specific doses (often similar to the native organic carbon content of sediment), the porewater concentrations and bioavailability of HOCs can be reduced between 70% and 99%. Furthermore, AC-moderated HOC sequestration often becomes more effective over time due to progressive mass transfer (Millward et al. 2005; Zimmerman et al. 2005; Werner et al. 2006; Sun et al. 2009; Ghosh et al. 2011; Cho et al. 2012).

Given these promising results, in situ sediment treatment involving the use of AC amendments is receiving increased attention among scientists, engineers, and regulatory agencies seeking to expand the list of remedial technologies and address documented or perceived limitations associated with traditional sediment remediation technologies. Based on the authors' review, AC is now the most widely used in situ sediment sequestration and immobilization amendment worldwide.

A previous review of the in situ AC remediation approach (Ghosh et al. 2011) reported the results of laboratory studies and early pilot-scale trials, summarized treatment mechanisms, highlighted promising opportunities to use in situ amendments to reduce contaminant exposure risks, and identified potential barriers for using this innovative technology. Another critical review by Janssen and Beckingham (2013) summarized the dependence of HOC bioaccumulation on AC dose and particle size, as well as the potential impacts of AC amendments on benthic communities (e.g., higher AC dose and smaller AC particle size further reduce bioaccumulation of HOCs but may induce stress in some organisms). This paper builds on these earlier reviews, focusing on design and implementation approaches involving the use of AC for in situ sediment treatment and summarizing key lessons learned.

DEMONSTRATING EFFICACY IN THE FIELD

Until recently, a primary challenge for full-scale in situ treatment remedies has been that most experience has emerged from laboratory and limited field pilot studies. Through 2013, however, more than 25 field-scale demonstrations or full-scale projects spanning a range of environmental conditions were completed or underway in the United States, Norway, and the Netherlands (Table 1 and Figure 1).

Among the more than 25 projects, field demonstrations in the lower Grasse River (Massena, NY, USA) and upper Canal Creek (Aberdeen, MD, USA) included the most comprehensive assessments and available documentation of the longer-term efficacy of the in situ AC remediation approach, although similar results have been reported for many of the other field projects. For this reason, the lower Grasse River and upper Canal Creek field demonstrations receive the greatest attention here, as summarized below.

Demonstration in lower Grasse River, Massena, New York

An AC pilot demonstration was conducted in the lower Grasse River as part of a program designed to evaluate available sediment cleanup options for the site. The demonstration study evaluated the effectiveness of AC as a means to sequester sediment polychlorinated biphenyls (PCBs) and reduce flux from sediments and uptake by biota.

The project began with laboratory studies and land-based equipment testing, and continued with field-scale testing of alternative placement methods. It culminated in a 2006 field demonstration of the most promising AC application and mixing methods to a 0.2-hectare pilot area of silt and fine sand sediments

Table?1. In situ sediment treatment using carbon-based sorbents (mainly AC): Summary of field-scale pilot demonstrations or full-scale projects

Site number (see Figure 1)	Year(s)	Location	Contaminant(s)	Application area (hectares)	Carbon-based amendment(s)	Delivery method(s)	Average water depth during delivery (m)	Enhancement(s)	Application equipment	Primary reference(s)
1	2004	Anacostia River, Washington, DC	PAHs	0.2	Coke Breeze	Geotextile mat	8	Armored cap	Crane	McDonough et?al. (2007)
2	2004, 2006	Hunters Point, San Francisco, CA	PCBs, PAHs	0.01	AC (slurry)	Direct placement	<1	Mechanical mixing (some areas)	Aquamog, slurry injection	Cho et?al. (2009 and 2012)
3	2006	Grasse River, Massena, NY	PCBs	0.2	AC (slurry)	Direct placement	5	Mechanical mixing (some areas)	Tine sled injection, tiller (with and without mixing)	Beckingham et?al. (2011); Alcoa (2007)
4	2006, 2008	Trondheim Harbor, Norway	PAHs, PCBs	0.1	AC (slurry)	Blended cover, direct placement	5	Armored cap (some areas)	Tremie, agricultural spreader	Cornelissen et?al. (2011)
5	2006	Spokane River, Spokane, WA	PCBs	1	Bituminous Coal Fines (slurry)	Direct placement	5	Armored cap	Mechanical bucket	Anchor QEA (2007 and 2009)
6	2009	De Veenkampen, Netherlands	Clean Sediment	<0.01	AC (slurry)	Direct placement	1	None	Laboratory rollerbank	Kupryianchyk et?al. (2012)
7	2009	Greenlandsfjords, Norway	Dioxins/Furans	5	AC (slurry)	Blended cover	30/100	None	Tremie from hopper dredge	Cornelissen et?al. (2012)
8	2009	Bailey Creek, Fort Eustis, VA	PCBs	0.03	AC (SediMite®)	Direct placement	1	None	Pneumatic spreader	Ghosh and Menzie (2012)
9	2010	Fiskerstrand Wharf, Ålesund, Norway	TBT	0.2	AC (slurry)	Blended cover	40	None	Tremie with biokalk	Eek and Schaanning (2012)
10	2010	Tittabawassee River, Midland, MI	Dioxins/Furans	0.1	AC (AquaGate™), Biochar	Blended cover	<1	None	Agricultural disc	Chai et?al. (2013)
11	2011	Upper Canal Creek, Aberdeen, MD	PCBs, Mercury	1	AC (SediMite®, AquaGate™, slurry)	Direct placement	<1	None	Pneumatic spreader, bark blower, hydroseeder	Bleiler et?al. (2013); Menzie et?al. (2014)
12	2011	Lower Canal Creek, Aberdeen, MD	Mercury, PCBs	0.04	AC (SediMite®)	Direct placement	1	None	Agricultural spreader	Menzie et?al. (2014)
13	2011 to 2016	Onondaga Lake, Syracuse, NY	Various Organic Chemicals	110	AC (slurry)	Blended cover	5	Armored cap	Hydraulic spreader	Parsons and Anchor QEA (2012)

(*Continued*)

Table 1. (*Continued*)

Site number (see Figure 1)	Year(s)	Location	Contaminant(s)	Application area (hectares)	Carbon-based amendment(s)	Delivery method(s)	Average water depth during delivery (m)	Enhancement(s)	Application equipment	Primary reference(s)
14	2011	South River, Waynesboro, VA	Mercury	0.02	Biochar (Cowboy Charcoal®)	Direct placement	<1	None	Pneumatic spreader	DuPont (2013)
15	2011	Sandefjord Harbor, Norway	PCBs, TBT, PAHs	0.02	AC (BioBlok[a])	Direct placement	30	None	Mechanical bucket	Lundh et?al. (2013)
16	2011	Kirkebukten, Bergen Harbor, Norway	PCBs, TBT	0.7	AC (BioBlok[a])	Direct placement	30	Armored cap (some areas)	Mechanical bucket	Hjartland et? al. (2013)
17	2012	Leirvik Sveis Shipyard, Sandefjord, Norway	PCBs, TBT, Various Metals	0.9	AC (BioBlok[a])	Direct placement	30	Armored cap (some areas)	Hydraulic spreader (up to 30-degree slopes)	Lundh et?al. (2013)
18	2012	Naudodden, Farsund, Norway	PCBs, PAHs, TBT, Various Metals	0.4	AC (BioBlok[a])	Direct placement	30	Armored cap, habitat layer	Mechanical bucket	Lundh et?al. (2013)
19	2012	Berry's Creek, East Rutherford, NJ	Mercury, PCBs	0.01	AC (SediMite®, granular)	Blended cover, direct placement	<1	None	Pneumatic spreader	USEPA (2013c)
20	2012	Puget Sound Shipyard, Bremerton, WA	PCBs, Mercury	0.2	AC (AquaGate™)	Direct placement	15	Armored cap	Telebelt® (under-pier)	Johnston et? al. (2013)
21	2012	Custom Plywood, Anacortes, WA	Dioxins/Furans	0.02	AC (SediMite®)	Blended cover, direct placement	8	None	Agricultural spreader	WDOE (2012)
22	2012	Duwamish Slip 4, Seattle, WA	PCBs	1	AC (slurry)	Blended cover	4	Armored cap	Mechanical bucket	City of Seattle (2012)
23	2013	Mirror Lake, Dover, DE	PCBs, Mercury	2	AC (SediMite®)	Direct placement	1	None	Telebelt® and air horn	DNREC (2013)
24	2013	Passaic River Mile 10.9, Newark, NJ	Dioxin/Furans, PCBs	2	AC (AquaGate™)	Blended cover	1	Armored cap	Telebelt®	In preparation
25	2013	Little Creek, Norfolk, VA	PCBs, various metals	1	AC (AquaGate™)	Direct placement	1	None	Pneumatic spreader (under-pier)	In preparation

AC, activated carbon; PAH, polynuclear aromatic hydrocarbon; PCB, polychlorinated biphenyl; TBT, tributyltin.
[a]BioBlok is licensed by AquaBlok®.

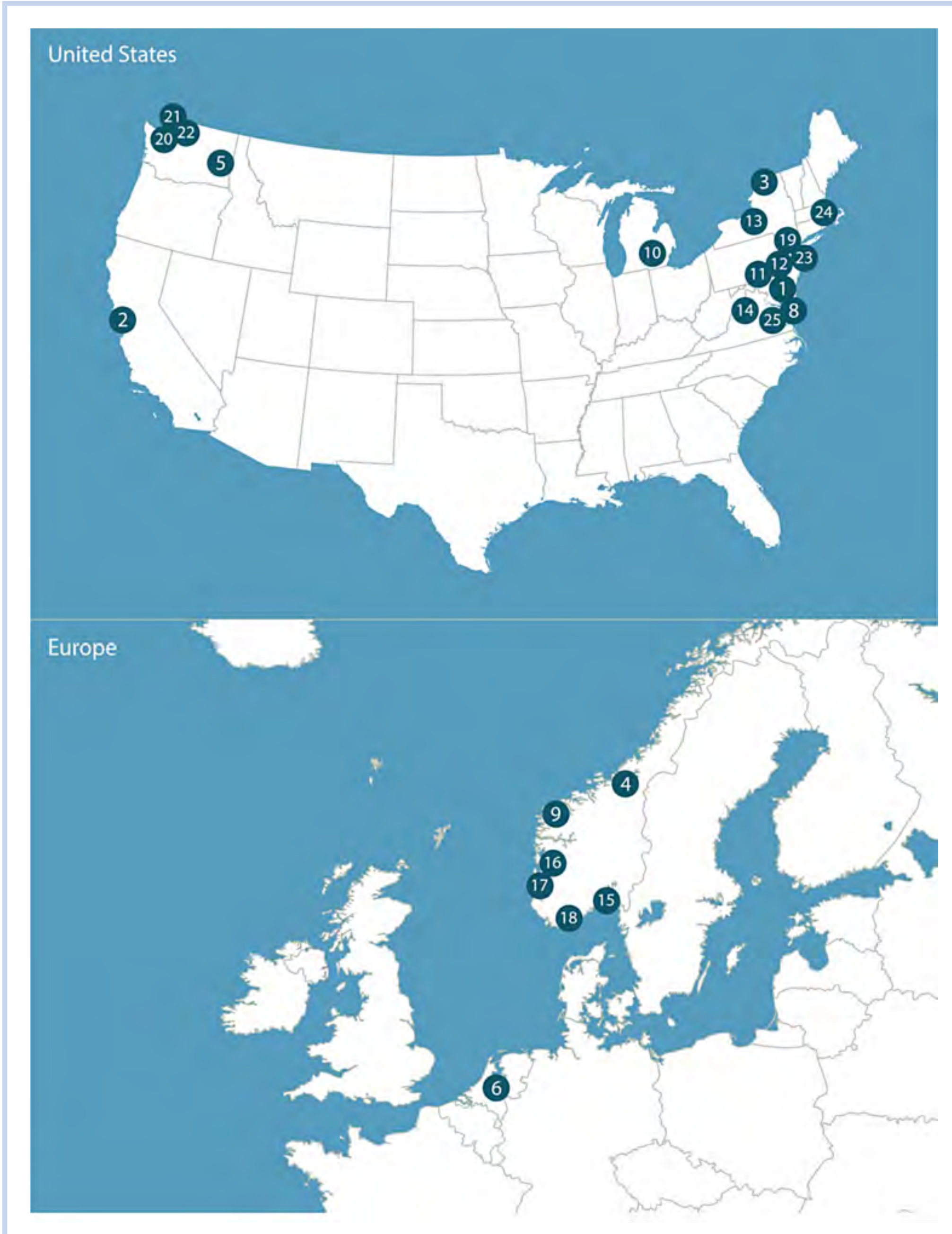

Figure 1. In situ sediment treatment field application sites (numbers refer to sites listed in Table 1).

at average water depths of approximately 5 meters (Alcoa 2007; Beckingham and Ghosh 2011).

The following application techniques were implemented in the Grasse River (Supplemental Figure S2):

- Applying (spraying) an AC slurry onto the submerged sediment surface and then mixing the material into near-surface sediments using a rototiller-type mechanical mixing unit (tiller)
- Injecting an AC slurry directly into near-surface sediments using a tine sled device (tine sled)
- Applying (spraying) an AC slurry onto the sediment surface within a temporary shroud enclosure, with no sediment mixing

All three application techniques successfully delivered the AC slurry onto or into surface sediments, and no detectable losses of AC to the water column or water quality impacts (e.g., turbidity monitored using instrumentation) were observed during placement (Alcoa 2007). A chemical oxidation method developed by Grossman and Ghosh (2009) was used to quantitatively confirm AC doses delivered onto or into sediment. This particular analytical method was used because typical total organic carbon and thermal (375 °C) oxidation methods were found to be imprecise and inaccurate, respectively, for AC analysis in sediment. Spraying the slurry onto the sediment successfully delivered AC to the sediment surface, and both the tiller with mixing and the tine sled applied all of the delivered AC into the 0- to 15-cm sediment layer. The tine sled application achieved more spatially (laterally) uniform doses, with an average AC concentration delivered to the 0- to 15-cm sediment layer of approximately $6.1 \pm 0.8\%$ AC (dry wt; ± 1 standard error around the mean based on core and surface grab sample data). This target (and applied) dose was approximately $1.5\times$ the native organic carbon content of the lower Grasse River. Cost comparisons of the different placement techniques indicate the tine sled unit would be a more cost-effective delivery method under full-scale deployment.

Detailed post-construction monitoring of the AC pilot area was performed in 2007, 2008, and 2009 (Beckingham and Ghosh 2011). Key findings are summarized below:

- AC addition decreased sediment porewater PCB concentrations, and reductions improved during the 3-year, post-placement monitoring period. Greater than 99% reductions in PCB aqueous equilibrium concentrations were observed during the third year of post-placement monitoring in plots where the AC dose in the 0- to 15-cm layer was 4% or greater (Figure 2), effectively demonstrating that PCB flux from sediments to surface water was almost completely contained.
- AC addition decreased PCB bioavailability as measured by in situ and ex situ bioaccumulation testing (using *Lumbriculus variegatus*). The overall decrease improved during the 3-year, post-placement monitoring period, with greater than 90% reductions observed during the third year of post-placement monitoring in plots where the AC dose in the 0- to 15-cm layer was greater than 4% (Figure 2).
- Benthic recolonization occurred rapidly after application and no changes to the benthic community structure or number of individuals were observed in AC amendment plots relative to background (Beckingham et al. 2013).
- In laboratory studies using site sediment, aquatic plants grew at a moderately reduced rate (approximately 25% less than controls) in sediment amended with a dose of greater than 5% AC. The reduced growth rate was likely attributable to nutrient dilution of the sediment (Beckingham et al. 2013).
- Although other project data (not shown) indicated the AC amendment slightly increased the erosion potential of sediments (although within the range of historical data for native sediments), all of the delivered AC remained in the sediments throughout the 3-year, post-placement monitoring period.
- Up to several centimeters of relatively clean, newly deposited sediment accumulated on the sediment surface in the pilot area over the 3-year, post-placement monitoring period. Passive sampling measurements revealed a downward flux of freely dissolved PCBs from the overlying water column into the AC amended sediments throughout the post-construction monitoring period. This suggested that the placed AC will continue to reduce PCB flux from sediments in the long term.

Demonstrations in upper Canal Creek, Aberdeen Proving Ground, Maryland

Two interrelated, pilot-scale, field demonstration projects were performed in 2011 to evaluate AC amendment additions to hydric soils at a tidal estuarine wetland in upper Canal Creek, at the Aberdeen Proving Ground, Maryland. (A third, separate treatment study was also carried out in the channelized portion of lower Canal Creek, but those results are only described minimally here.)

The first demonstration pilot (Menzie et al. 2014) evaluated in situ treatment with SediMite® pellets, a proprietary system for delivering powdered AC treatment materials with a weighting agent and an inert binder (Ghosh and Menzie 2010 2012). The second demonstration pilot (Bleiler et al. 2013) evaluated two different powdered AC-bearing treatment materials: AquaGate + PAC™ (AquaGate) and a slurry containing AC. The proprietary AquaGate product typically includes a dense aggregate core, along with clay-sized materials, polymers, and powdered AC additives. For both field demonstrations and all AC-bearing materials, the objective was to reduce PCB exposure to invertebrates living on or within surface sediments of the wetland area and thus reduce exposure to wildlife that might feed on these invertebrates.

All three AC-containing treatment materials for these pilot projects were applied onto the surface of the wetland and creek sediments during seasonal and tidal conditions with little or no overlying water. A total of 20 plots (each 8×78 meters) were

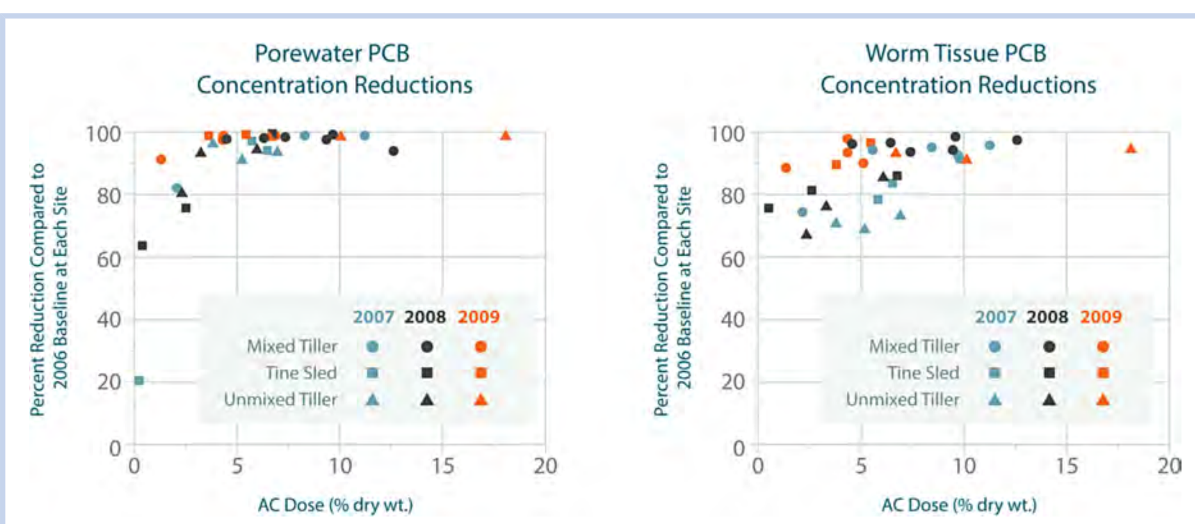

Figure 2. Reductions in porewater and worm tissue PCB concentrations at lower Grasse River, NY.

used for the demonstration projects; sampling was conducted prior to application and at 6 and 10 months following application. Performance measurements used in one or both of the pilot projects included porewater and macroinvertebrate tissue PCB concentrations; phytotoxicity bioassays; ecological community abundance, diversity, and growth surveys; and nutrient uptake studies. Treatment efficacy was evaluated by comparing pre- versus post-treatment metrics and by evaluating treated plots relative to control (no action) and conventional sand cap plots.

The three treatment materials—SediMite®, AquaGate, and AC in a slurry—were applied using a pneumatic spreader, a bark blower, and a hydroseeder, respectively (Supplemental Figure S3). Figure S3 also shows a barge-mounted agricultural spreader that was used to demonstrate delivery of SediMite® to a portion of lower Canal Creek.

For both field demonstrations and all AC-bearing materials, the treatment goal was to achieve a 3% to 7% (dry wt) AC concentration in wetland surface sediment, which was operationally defined as the upper 10 cm (SediMite® studies) and 15 cm (AquaGate and slurry studies). Because the materials contained different amounts of AC, the applications differed in target thickness on the wetland surface. SediMite® contains approximately 50% AC by dry weight, so the target dose of 5% in the top 10 cm of sediment resulted in a target amendment layer thickness of roughly 0.7 cm. In contrast, AquaGate contained a coating of 5% powdered AC and was thus applied as a thicker 3-cm to 5-cm target layer over the sediment. The slurry system delivered roughly 0.2 cm to 0.5 cm of concentrated AC on the surface of the marsh. All of the treatments relied on natural processes (bioturbation, sediment deposition, and other physical processes) to mix AC placed onto the sediment surface into the wetland and creek sediment over time (see post-construction monitoring discussion below).

The AC amendments were applied effectively onto wetland and creek sediments in all of the applications. Measurements made over time indicated that close to 100% of the AC was retained within the plots, but vertical mixing into native wetland sediments via natural processes was slower than originally anticipated. As a result of low bioturbation rates, AC applied in more concentrated forms (i.e., as SediMite® and as AC in a slurry) remained at concentrations greater than the target dose of 5% in the upper 2 cm of the wetland sediment layer 10 months following application (Supplemental Figure S4). During the 10-month, post-application monitoring period, AC was incorporated into the biologically active zone largely from localized root elongation processes (Bleiler et al. 2013). Based on the two post-application monitoring rounds, approximately 60% of the recovered AC was found in the top 2 cm of sediment, whereas the remaining 40% penetrated mostly in the 2- to 5-cm depth interval. It is expected that further incorporation of the AC into the deeper layers of sediment will occur slowly over time via natural mixing processes and deposition of new sediment and organic matter.

The effectiveness of the AC amendments applied to the upper Canal Creek wetlands was assessed by measuring reductions in PCB concentrations in porewater (in situ measurements) and macroinvertebrate tissue (ex situ bioaccumulation testing). PCB concentrations exhibited a large spatial variability (1 order of magnitude) and vertical variability (up to 2 orders of magnitude within a sediment depth of 20 cm) in sediments across the plots, which was a site condition before the AC was applied. This finding posed some challenges in interpreting data and was therefore taken into account when evaluating other metrics. The findings of the upper Canal Creek demonstration pilot are reported in detail in Menzie et al. (2014) and Bleiler et al. (2013).

Regardless of the above challenges, all AC-treated wetland plots showed reduced PCB bioavailability as measured by reductions in both benthic organism tissue and porewater concentrations during the post-application monitoring period. In addition, no significant phytotoxicity or changes in species abundance, richness or diversity, vegetative cover, or shoot weight or length were observed between the AC treatment and control plots. Furthermore, plant nutrient uptake in the AC treatment plots was not significantly lower than control plots. Although the overall findings of these pilot projects suggest that adding AC can sequester PCBs in wetland sediments, more monitoring will take place given the slow mixing of the placed AC into the underlying wetland and creek sediments.

The lower Grasse River and upper Canal Creek projects, along with the other field-scale projects summarized in Table 1, collectively demonstrate the efficacy of full-scale in situ sediment sequestration and immobilization treatment technologies. Such efforts reduce the bioavailability and mobility of several HOC and other contaminants, including PCBs, polynuclear aromatic hydrocarbons, dioxins and furans, tributyltin, methylmercury, and similar chemicals. Results from these field applications indicate that in situ treatment of contaminants can reduce risks rapidly by addressing key exposures (e.g., bioaccumulation in invertebrates), often becoming more effective over time due to progressive mass transfer.

APPLICATION METHODS AND EXAMPLES

The AC application projects summarized in Table 1 involved placing amendments using several options that fall into two broad categories (Figure 3):

1) Direct application of a thin layer of sorptive, carbon-based amendments (which potentially incorporates weighting or binding materials) onto the surface sediment, with or without initial mixing
2) Incorporating amendments into a pre-mixed, blended cover material of clean sand or sediment, which is also applied onto the sediment surface

Although these approaches have several differences, the ultimate goal of both is to reduce exposure of benthic organisms to HOCs in sediment and reduce HOC flux from sediment into water (Figure 3). Under either approach, the applied AC may mix eventually throughout the biologically active layer via bioturbation. Application methods are described further in the next sections.

Direct application method

Using this approach, the bioavailability of HOCs in surface sediments is reduced by directly applying a strong carbon-based sorbent such as AC. At the lower Grasse River, upper Canal Creek, and many other field demonstration or full-scale projects (Table 1), AC amendment was applied successfully using several methods with or without mixing, weighting agents, inert binders, or other proprietary systems. The specific application method was optimized to site-specific conditions.

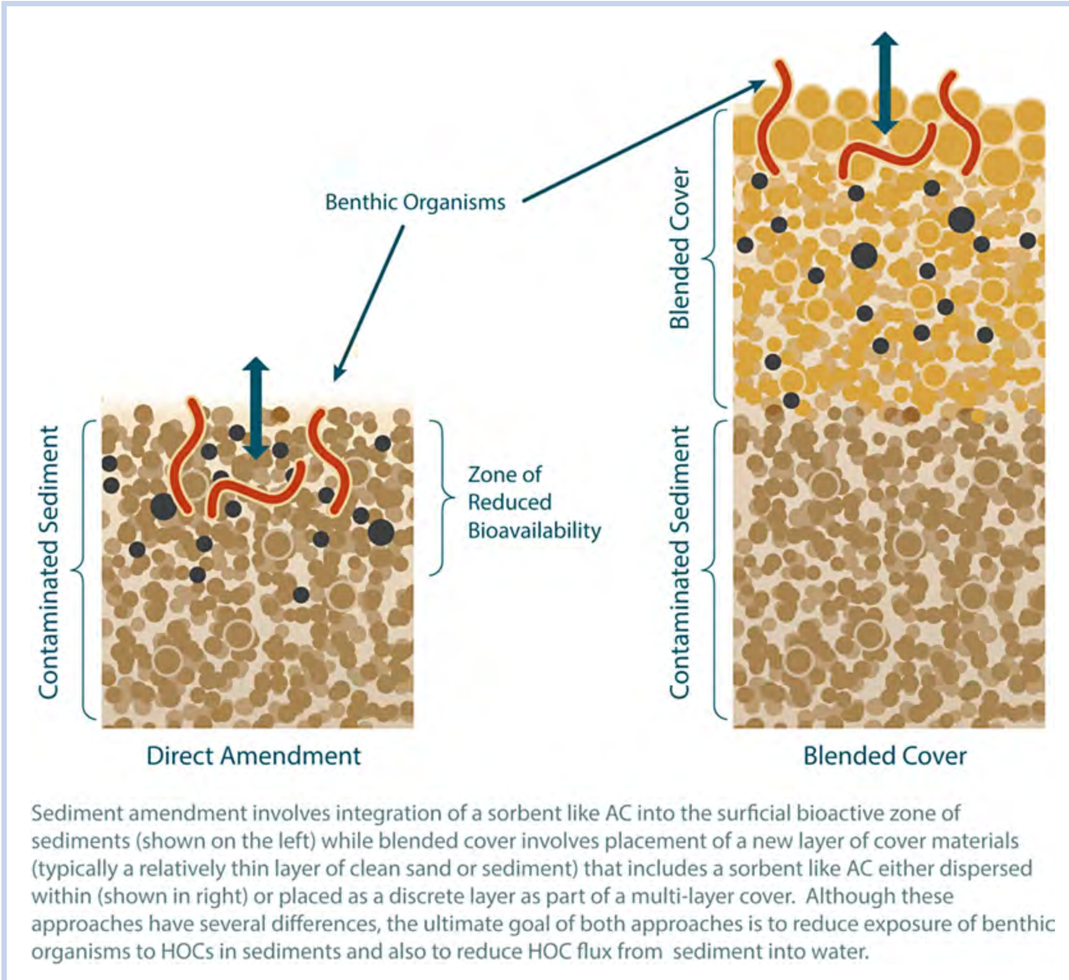

Figure 3. Direct amendment versus blended cover application methods for in situ sorbent application.

Adding weighting agents or inert binders can often improve the placement accuracy of finer-grained AC materials.

When the amendment introduced consists primarily of the sorbent, the direct application approach introduces minimal new material (an advantage), with little or no change in bathymetry or ecological habitat including the sediment's physical and mineralogical characteristics. Applying amendment to sediment surfaces also allows for some capacity to treat new contaminated sediments that may be deposited after constructing the remedy. This approach may have particular advantages at ecologically sensitive sites, where maintaining water depth is critical, and also where the potential for erosion is low.

The Delaware Department of Natural Resources and Environmental Control conceived and funded the first full-scale example of direct placement of AC in the United States, which was implemented in Mirror Lake, a reservoir on the St. Jones River in Dover, Delaware (Table 1; Site 23). The sediment cleanup remedy at this site aimed to enhance the sorption capacity of native sediments in the lake, such that PCB bioavailability to the food chain is reduced without greatly altering the existing sediment bed. The remedy included placing SediMite® over an approximate 2-hectare area in the lake and river, along with integrated habitat restoration (DNREC 2013).

Placing AC at Mirror Lake was performed in the fall of 2013 using two application methods (Supplemental Figure S5): a Telebelt® application for the most accessible parts of the lake and an air horn device to pneumatically deliver SediMite® from a boat and along nearshore areas. Heavy equipment could not be deployed in the lake due to shallow water depth (averaging roughly 1 meter), as well as soft bottom sediments. The SediMite® application was completed safely in approximately 2 weeks. The target (and measured) thickness of the applied SediMite® material was approximately 0.7 cm, with the material expected to integrate naturally into the surficial sediment over time. Grab samples (13 stations) were collected from the top 10 cm of sediment in the lake 2 weeks after application to measure AC based on a method described in Grossman and Ghosh (2009). Applying SediMite® achieved an average AC dose of $4.3 \pm 1.6\%$ (Supplemental Figure S6).

Blended cover application method

The blended cover application method is a variation of the enhanced natural recovery remedy described by the US Environmental Protection Agency (USEPA 2005). In this approach, the carbon-based sorbent material is premixed with relatively inert materials such as clean sand or sediment and placed onto the contaminated sediment surface. Although this approach involves introducing materials in addition to the sorbent, it may have advantages at sites where a more spatially (vertically and laterally) uniform application of AC to the sediment surface is desired (because the AC can be mixed more thoroughly with the sand or sediment) or where more rapid control of HOC flux is desired.

Laboratory experiments and modeling studies (Murphy et al. 2006; Eek et al. 2008; Gidley et al. 2012), as well as field demonstrations (McDonough et al. 2007; Cornelissen et al. 2011, 2012) have confirmed the effectiveness of the blended cover application approach in reducing flux of mobile HOCs. At sites where additional isolation or erosion protection of underlying contaminated sediments may be needed, a related but separate option is to apply the sorbent as a layer within a conventional armored isolation cap. This paper, however, does not review either conventional or reactive isolation caps as defined by the USEPA (2005).

A full-scale example of blended AC application began in 2012 at Onondaga Lake, located in Syracuse, New York. The sediment cleanup remedy included placing bulk granular AC (GAC) blended with clean sand over approximately 110 hectares of lake sediments, along with related armored capping, dredging, and habitat restoration actions (NYSDEC and USEPA 2005; Parsons and Anchor QEA 2012). Full-scale implementation began following a successful field demonstration in fall 2011 and is currently scheduled to be completed in 2016.

Placing the blended GAC material in Onondaga Lake is being accomplished using a hydraulic spreading unit with advanced monitoring and control systems capable of placing approximately 100 cubic meters per hour of material in 6-meter-wide lanes (Figure 4). Granular AC amendment is mixed with sand and hydraulically transported and spread over sediment (average water depth of approximately 5 meters) through a diffuser barge. The GAC is presoaked for at least 8 hr prior to hydraulic mixing with the sand, to improve the settlement of the GAC through the water column. The spreader barge is equipped with an energy diffuser to distribute the blended materials evenly. The spreader barge incorporates electronic position tracking equipment and software so that the location of material placement can be tracked in real time. The spreader barge is also equipped with instruments for measuring the density of the slurry and the flow rates, which together provide the instantaneous production rate of the blended material being placed. Granular AC application rates are also tightly controlled and monitored using peristaltic metering pumps and a slurry density flow meter. The land-based slurry feed system is metered to the desired GAC dose.

Through the first 2 years of the 5-year construction project, the blended GAC material was placed in Onondaga Lake without any detectable losses to the water column. Verifying GAC placement was performed using both in situ catch pans located on the sediment surface prior to placement, as well as cores collected after placement. Results of these verifications demonstrated that the GAC was placed uniformly both horizontally and vertically within the sand layer applied to the lake (Supplemental Figure S7).

Figure 4. Hydraulic spreading application unit at Onondaga Lake, Syracuse, NY.

SITE EVALUATION AND DESIGN CONSIDERATIONS

The more than 25 field-scale demonstrations or full-scale projects performed through 2013 span a range of application methods and environmental conditions (including marine, brackish, and freshwater sites; tidal wetlands and mudflats; deep depths; steep slopes; under piers; and moving water [Table 1]). Collectively, these projects demonstrate the efficacy of in situ sediment treatment using sorptive, carbon-based amendments, particularly AC. As a result, in situ sediment treatment using AC is ready for full-scale application at a range of sites, subject to careful site-specific design analyses, generally as outlined in the next paragraphs.

To determine if site conditions are favorable for AC amendment, relatively simple bench testing of AC amendments can be performed by mechanically mixing AC into the sediments and performing straightforward porewater or bioaccumulation testing (e.g., Sun and Ghosh 2007). Short-term bench testing performed in this manner can rapidly identify sediment sites that are amenable to sediment treatment with AC and can be coupled with focused modeling or column studies to evaluate HOC behavior associated with groundwater flux. Bench testing can also be used to optimize AC materials (e.g., grain size or porosity) and dosing based on site-specific conditions. (Note that at most of the sites listed in Table 1, optimal AC doses were similar to the native organic carbon content of sediment.)

Although much has been learned to date, additional focused field-scale demonstrations may be particularly helpful to evaluate certain site-specific HOCs such as dioxins, furans, and methylmercury for which treatment effectiveness has been either variable or slow to develop (i.e., after the AC is mixed in) and in environments where sorptive carbon-based amendments have not yet been piloted (e.g., high-energy, erosion-prone locations). It is also important to note that at some sites, AC application may not provide additional protection compared to traditional sediment cleanup technologies. For example, mixing AC into a blended cover at Grenlandsfjords, Norway resulted in only marginal additional dioxin and furan flux reductions at 9 and 20 months compared with unamended clean sand or sediment cover materials, attributable in part to relatively slow sediment-to-AC transfer rates for large molecular volume dioxins and furans (Cornelissen et al. 2012; Eek and Schaanning 2012).

Based on a critical review of the results of the field-scale projects listed in Table 1, specific-site and sediment characteristics can reduce the effectiveness of AC application compared to other potential sediment cleanup technologies. These characteristics include (but are not likely limited to) relatively high native concentrations of black carbonaceous particles and slow sediment-to-AC transfer rates for relatively large molecular volume HOCs (Choi et al. 2014). Properly accounting for these and factors such as erosional forces and mixing or bioturbation in site-specific AC application design is necessary to ensure the effectiveness of the in situ remedial approach.

Experimental, modeling, and long-term monitoring lines of evidence from the case studies summarized in Table 1 have all confirmed that the effectiveness of AC applications increases over time at sites where there is not a significant flux from the underlying sediment to the surface. In many settings, full treatment effectiveness of AC amendments is achieved years after installation (e.g., Werner et al. 2006; Cho et al. 2012). The delay can be caused by (among other factors) the heterogeneity of AC distribution (even on a small scale), particularly at sites with relatively low bioturbation rates, as well as progressive mass transfer (Figure 5).

Site-specific evaluations of natural sediment deposition and bioturbation rates (as well as ongoing contaminant sources) and their effect on AC mixing and resultant restoration time frames are important design factors in developing appropriate site-specific in situ treatment strategies. Rates of natural sediment deposition and bioturbation-induced mixing of AC into the biologically active zone vary widely between sediment environments. For example, surface sediment bioturbation rates have been shown to vary more than 2 orders of magnitude between sediment environments, with relatively lower rates in wetlands and offshore sediments and relatively higher rates in productive estuaries and lakes (e.g., Officer and Lynch 1989; Wheatcroft and Martin 1996; Sandnes et al. 2000; Parsons and Anchor QEA 2012; Menzie et al. 2014). If relatively slow rates of natural deposition and mixing are anticipated, applying AC directly could be staggered over multiple applications to incorporate the amendment more evenly into the depositing sediments, albeit with potential cost implications.

As the USEPA (2005), NRC (2007), Bridges et al. (2010), ITRC (2014), and others have emphasized, the effectiveness of all sediment cleanup technologies depends significantly on sediment- and site-specific conditions. For example, resuspension and release of sediment contaminants occurs during environmental dredging, particularly at sites with debris and other difficult dredging conditions (Patmont et al. 2013). Optimizing risk management at contaminated sediment sites can often be informed by comparative evaluations of sediment cleanup technologies applied to site-specific conditions, considering quantitative estimates of risk reduction, risk of remedy, and remedy cost (e.g., Bridges et al. 2012). A hypothetical comparative risk reduction evaluation is presented in Figure 6 and highlights some of the short- and long-term tradeoffs that can occur between different sediment remediation technologies. Consistent with the example presented in Figure 6, at many sites, AC placement can achieve risk reductions similar to conventional capping but at a lower cost (see below), and may also provide better overall risk reduction than environmental dredging. Although Figure 6 presents a relatively common sediment remedial alternatives evaluation scenario in North America, it is important to note that site-specific conditions will result in varying risk reduction outcomes from alternative sediment remedies.

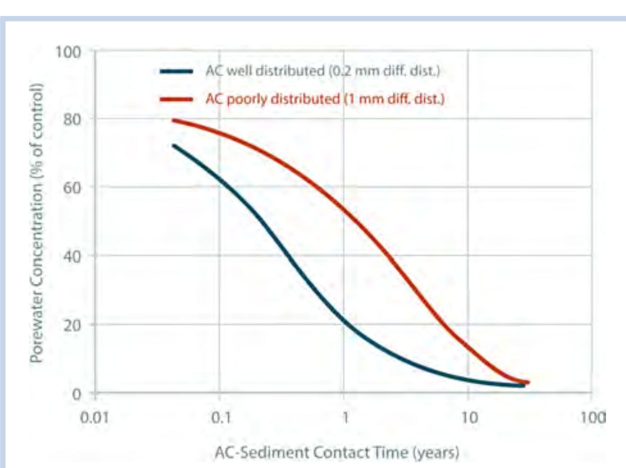

Figure 5. Model simulations of porewater PCB concentration reductions with different mixing scenarios (adapted from Cho et al. 2012).

POTENTIAL NEGATIVE ECOLOGICAL IMPACTS

The acceptability of any sediment remediation option will depend on whether the benefits of the approach outweigh potential adverse environmental or ecological impacts, compared to other options. Because in situ treatment technologies involve adding a new material to sediments, in situ remedies have the potential to impact the native benthic community and vegetation, at least temporarily. A recent review by Janssen and Beckingham (2013) found that impacts to benthic organisms resulting from AC exposure were observed in one-fifth of 82 tests (primarily laboratory studies). Importantly, community effects have been observed more rarely in AC field pilot demonstrations compared to laboratory tests and often diminish within 1 or 2 years following placement (Cornelissen et al. 2011; Kupryianchyk et al. 2012), particularly in depositional environments where new (typically cleaner) sediment continues to deposit over time.

Although applying relatively higher AC doses or smaller AC particle sizes provide greater bioaccumulation reductions of HOCs, higher doses and smaller particle size may induce greater stress in some organisms (Beckingham et al. 2013). Negative impacts to benthic macroinvertebrates and aquatic plants resulting from adding AC, particularly at relatively high doses, may be attributable to nutrient reductions associated with AC amendment.

Although the available dose–dependent effects data for AC are not comprehensive, field trials and experimental studies suggest that potential negative ecological effects can be minimized by maintaining finer-grained AC doses below

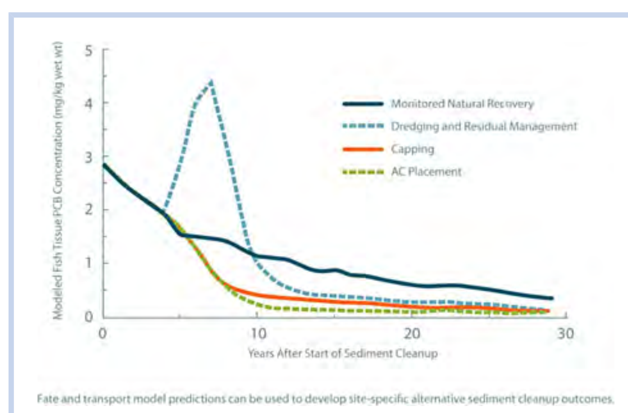

Figure 6. Hypothetical comparative net risk reduction of alternative sediment remedies. Example presented for illustrative purposes using the following fate and transport model input assumptions: average environmental dredge production rate of 400 m^3 per day and release of 3% of the PCB mass dredged (Patmont et al. 2013); average water flow through the cleanup area of 500 m^3 per second; implementation of effective upstream source controls; net sedimentation rate of 0.1 cm per year; and typical PCB mobility and bioaccumulation parameters.

Table 2. Summary of low- and high-range unit costs of AC application[a]

Component	Low-range Unit Cost	High-range Unit Cost
Activated Carbon[b]	$50,000/hectare	$100,000/hectare
Facilitating AC Placement Using Binder/Weighting Agents[c]	$0/hectare	$70,000/hectare
Facilitating AC Placement by Blending with Sediment or Sand[c]	$0/hectare	$100,000/hectare
Field Placement	$30,000/hectare	$200,000/hectare
Long-term Monitoring	$20,000/hectare	$100,000/hectare[d]
Total	$100,000/hectare	$500,000/hectare

[a]Estimated costs for a 4 percent AC dose (dry weight basis) over the top 10-cm sediment layer at a 5-hectare site.
[b]Powdered activated carbon (PAC) and/or granular activated carbon (GAC), depending on site-specific designs.
[c]To facilitate AC placement, binder or weighting agent amendments such as SediMite® or AquaGateTM, or clean sediment or sand (but typically not both) may be required in some applications depending on site-specific conditions and designs.
[d]High-end monitoring cost of $100,000 per hectare reflects prior pilot projects and likely overestimates costs for full-scale remedy implementation.

approximately 5% (dry wt basis; e.g., see discussion of the lower Grasse River AC demonstration). Similar to the net risk reduction comparisons summarized in Figure 6, the positive effects of reduced bioaccumulation of HOCs need to be balanced against potential negative short-term impacts. In addition, site-specific outcomes from in situ AC applications should be compared with outcomes resulting from other remediation approaches such as dredging and conventional capping, which are often greater than those resulting from in situ treatment.

RELATIVE SUSTAINABILITY OF DIFFERENT CARBON AMENDMENTS

Although amendments produced from different carbon source materials often exhibit similar effectiveness and negative ecological effects, different types of carbon amendments have different sustainability attributes. For example, life cycle analyses have demonstrated that AC produced from anthracite coal is less sustainable than AC produced from biomass feedstock (Sparrevik et al. 2011; e.g., agricultural residues), even though anthracite-derived AC may bind HOCs very effectively (Josefsson et al. 2012). One important positive effect of biomass AC related to sustainability is that its carbon is sequestered and removed from the global carbon cycle (Sparrevik et al. 2011). Even better sustainability outcomes can result from using non-activated pyrolyzed carbon, or “biochar” (Ahmad et al. 2014), because considerable amounts of energy are required for the activation process. However, the sorption capacity of biochars for many HOCs is more than an order of magnitude lower than AC (Gomez-Eyles et al. 2013).

COST

Based on a critical review of the field-scale projects listed in Table 1 for which adequate cost information was available, we summarized approximate low- and high-range unit costs for a full-scale AC application to a hypothetical 5-hectare sediment cleanup site. Cost summaries for the primary implementation components, not all of which may be needed at a particular site, are summarized in Table 2. Based on this summary, AC application is often likely to be less costly than either traditional dredging or capping approaches. Again, site-specific conditions can result in varying cost outcomes from alternative sediment remedies.

CONCLUSION

In situ sediment treatment using AC can rapidly address key exposures (e.g., bioaccumulation in invertebrates and fish), often becoming more effective over time due to progressive mass transfer. Due to its relatively large surface area, pore volume, and absorptive capacity, AC has a decades-long track record of effective use as a stable treatment medium in water, wastewater, and air. As such, AC is well suited for in situ sequestration and immobilization of HOCs in various sediment environments.

When designed correctly to address site-specific conditions, controlled (accurate and spatially uniform) placement of AC-bearing treatment materials has been demonstrated using a range of conventional construction equipment and delivery mechanisms and in a wide range of aquatic environments (Table 1), including wetlands. When contaminated sediments are present in unstable environments, traditional capping or dredging remedies might be the preferred option. Depending on sediment and site conditions, however, using AC can achieve short-term risk reduction similar to conventional capping and better overall risk reduction than environmental dredging, with lower costs and environmental impacts than traditional sediment cleanup technologies.

With a growing international emphasis on sustainability, in situ sediment treatment remedies offer an opportunity to realize significant environmental benefits, while avoiding the environmental impacts often associated with more invasive sediment cleanup technologies. Less invasive remediation strategies—such as treatment using in situ AC applications—are also typically far less disruptive to communities and stakeholders than dredging or conventional capping remedies. Important environmental, economic, and other sustainability issues can be associated with in situ sediment treatment, such as low-impact reduction of the bioavailable or mobile fractions of sediment contaminants through sequestration, improved recovery time frames, and reduced energy use and emissions (e.g., carbon; ITRC 2014).

Proper site-specific balancing of the potential benefits, negative ecological effects, and costs of in situ treatment relative to other sediment cleanup technologies is important to applying this approach successfully at full-scale. As discussed in USEPA (2005) and ITRC (2014), at most sites, a combination of sediment cleanup technologies applied to specific zones within the sediment cleanup site will result in a remedy that achieves long-term protection while minimizing short-term negative impacts and achieving greater cost effectiveness. It is evident from the extensive experimental studies and field-scale projects presented here that when applied correctly, in situ treatment of sediment HOCs using sorptive, AC-bearing materials has progressed from an innovative sediment remediation approach to a proven, reliable technology. Indeed, it is one that is ready for full-scale remedial application in a range of aquatic sites.

SUPPLEMENTAL DATA

Figure'S1. Simplified food chain model of in situ treatment.

Figure'S2. Pilot area and tine sled or tiller application units at lower Grasse River, NY.

Figure'S3. Dry broadcasting and slurry spray applications, Canal Creek, Aberdeen Proving Ground, MD.

Figure'S4. Vertical distribution of AC in wetland sediments at Canal Creek, Aberdeen Proving Ground, MD.

Figure'S5. SediMite® delivery at Mirror Lake, Dover, DE.

Figure'S6. Post-placement surface sediment AC concentrations at Mirror Lake, Dover, DE.

Figure'S7. Applied versus measured AC dose at Onondaga Lake, Syracuse, NY.

Acknowledgment—The authors gratefully acknowledge the advice and support of the Sediment Management Work Group, whose members (including Steve Nadeau, Larry McShea, Bill Hague, Will Gala, Steve Brown, and Joe Chai, among others) provided encouragement and constructive reviews of early drafts of this paper.

Disclaimer—Views or opinions expressed in this paper do not necessarily reflect the policy or guidance of the USEPA

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<u>Attachment E</u>

Environmental Science & Technology

FEATURE

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In-situ Sorbent Amendments: A New Direction in Contaminated Sediment Management†

Upal Ghosh*

University of Maryland Baltimore County, Baltimore, Maryland 21250, United States

Richard G. Luthy

Stanford University, Stanford, California, United States

Gerard Cornelissen

Norwegian Geotechnical Institute, Oslo, Norway, University of Life Sciences, Ås, Norway; Stockholm University, Stockholm, Sweden

David Werner

Newcastle University, Newcastle upon Tyne, United Kingdom

Charles A. Menzie

Exponent, Alexandria, Virginia, United States

S Supporting Information

Aquatic sediments form the ultimate repositories of past and ongoing discharges of hydrophobic organic compounds (HOCs) such as polychlorinated biphenyls (PCBs), many pesticides, and dioxins, as well as mercury (Hg) and methylmercury (MeHg). These sediment-bound pollutants serve as long-term exposure sources to aquatic ecosystems. Approximately 10% of the sediment underlying the United States' surface water is sufficiently contaminated with toxic pollutants to pose potential risks to fish and fish-eating wildlife and humans.[1] Remediation of contaminated sediments remains a technological challenge. Traditional approaches do not always achieve risk reduction goals for human health and ecosystem protection and can even be destructive for natural resources. Though removal of contaminated sediment by dredging and disposal in a secure landfill can be effective under certain conditions, a recent study by the National Research Council found a wide range of outcomes.[2] Among the problems with dredging are unfavorable site conditions, resuspension of contaminated sediment into the water column, and contaminated sediment residuals. While capping contaminated sediment with clean sand may be a viable remedial option at some sites, often the alteration of sediment bathymetry may not be acceptable and the control of contaminant transport through the cap can be a challenge. In addition, both dredging and conventional capping result in the destruction of existing benthic ecosystems. Therefore, development of new techniques offering greater flexibility in contaminated sediment management and avoiding some of the problems with conventional dredging and capping is highly desirable.

This feature article summarizes research by several groups in the U.S. and Europe to develop a novel approach for in situ sediment remediation that minimizes or eliminates some of the problems with traditional technologies. The efforts involve introducing sorbent amendments into contaminated sediments that alter sediment geochemistry, increase contaminant binding, and reduce contaminant exposure risks to people and the environment. We present here a description of recently concluded laboratory studies and a brief outline of ongoing pilot-scale trials, field challenges, regulatory issues, and further research needs.

BIOAVAILABILITY OF SEDIMENT-BOUND LEGACY CONTAMINANTS

Sediment HOCs can be taken up by aquatic or benthic organisms through ingestion and dermal absorption, and subsequently passed on to higher organisms and humans. For both of these pathways, the uptake depends on the bioavailability of contaminants in sediment, which is determined by how strongly the contaminants are bound to the sediment particles.[3,4] Strong binding in the sediment matrix reduces contaminant bioavailability to organisms. Work in the last two decades has improved our understanding of how sediment geochemistry controls contaminant bioavailability. For example, black carbonaceous particles in sediments such as soot, coal, and charcoal very strongly bind HOCs, and their presence in sediments (both natural and anthropogenic) reduces exposure and risk,[5,6] often by one order of magnitude or more compared to natural organic matter.

CONTAMINANT SEQUESTRATION BY ACTIVE AMENDMENTS

"Natural" contaminant sequestration in native carbonaceous particles can be greatly enhanced by the addition of clean,

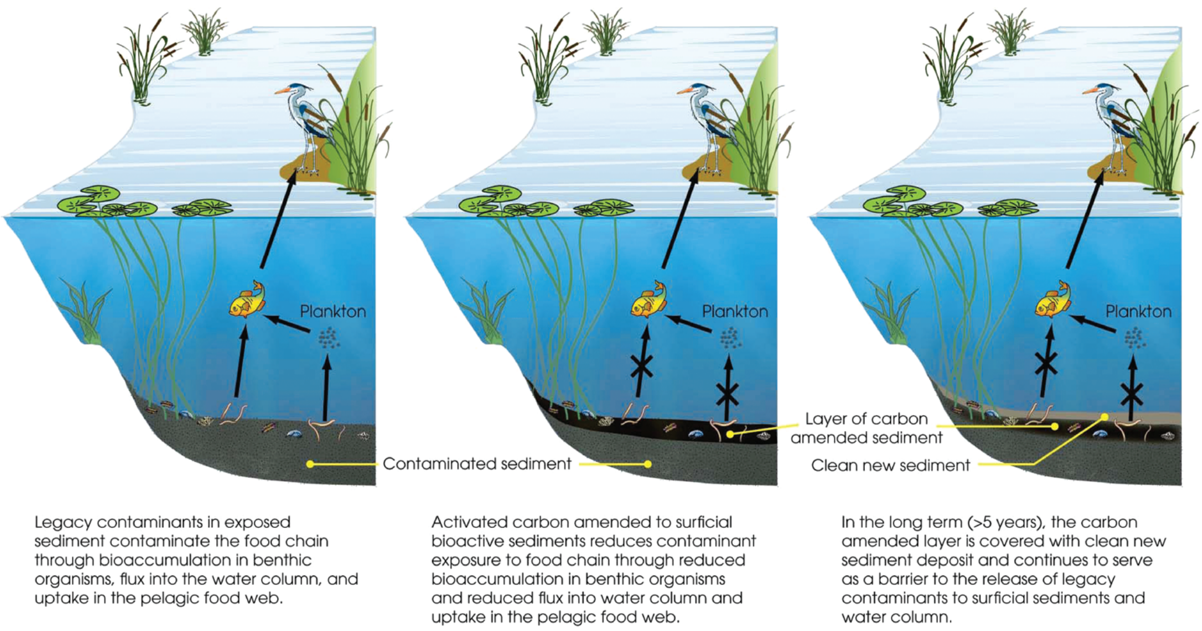

Figure 1. Conceptual model of how sorbent amendment of sediment reduces contaminant exposure pathways of benthic organism accumulation and flux from the sediment bed.

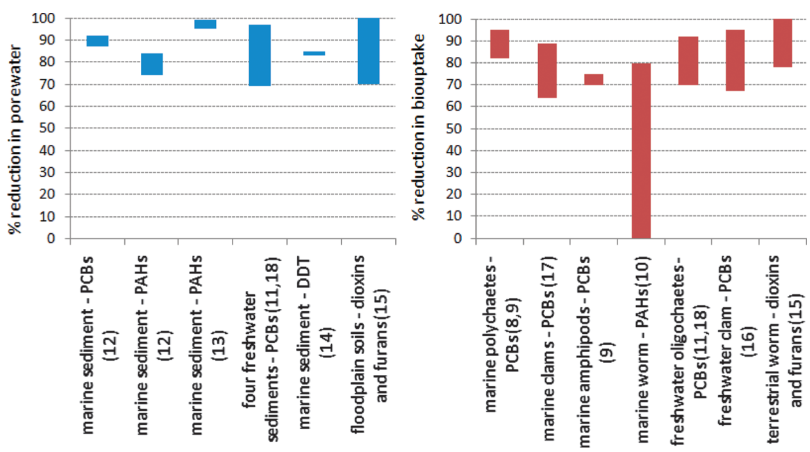

Figure 2. Percent reduction ranges of aqueous equilibrium concentration and contaminant biouptake in different laboratory studies of activated carbon amendment to sediments and soils from the field. These studies range from freshwater to marine sediments and cover a wide range of benthic organisms. The dose of activated carbon used in these laboratory experiments typically ranged from 1 to 5% by dry sediment weight (*29*).

manufactured carbonaceous materials into sediments, such as activated carbon (AC). AC is produced from coal or biomass feedstock and treated at high temperature to produce a highly porous structure with great sorption capacity. Activated carbons have been used widely for drinking water purification and human poisoning abatement. McLeod et al [7] showed in clam particle feeding studies that the biouptake of a tetrachloro-PCB in the gut was only 1−2% for AC-sorbed PCBs, compared to 90% for diatom-sorbed ones. As illustrated in Figure 1, amending or thin-capping the bioactive surface layer of sediment with AC will transfer contaminants from the sediment to the strongly binding AC particles, reducing bioavailability to benthic organisms and contaminant flux into the water column, and thus accumulation in the aquatic food-chain. Sediment turnover by benthic organisms and other natural mixing processes can further incorporate the added AC into deeper or newly depositing sediment layers.[11] In depositional sediment environments, where legacy contaminants are often found, over time new clean sediment can cover the AC-treated sediment layer (Figure 1).

Laboratory tests with contaminated sediment show proof-of-concept through reductions in HOC bioavailability (Figure 2). These studies evaluated HOC bioavailability through measurement of equilibrium aqueous concentration and biouptake in a range of benthic organisms. The study sediments were all field-collected and had aged for decades in freshwater or marine environments. HOC concentrations in sediment porewater provide a useful assessment of the potential sediment-to-water flux, especially when legacy contaminated sediments are the primary pollution source. Sediment porewater concentration is also predictive of HOC biouptake in benthic organisms.[19] Tests with a range of field sediments showed that AC amendment in the range of 1−5% reduces equilibrium porewater concentration

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of PCBs, PAHs, DDT, dioxins, and furans in the range of 70−99%, thus reducing the driving force for the diffusive flux of HOCs into the water column and transfer into organisms. Most of the studies using benthic organisms show a reduction of biouptake of HOCs in the range of 70−90% compared to untreated control sediment (Figure 2).

Recent work on metal-contaminated sediments demonstrated reduced biouptake of cadmium (Cd) [20] and Hg/MeHg [21] after amendment of AC and thiol-functionalized silica into sediments. Significant reductions in Hg from water may be feasible with polysulfide-rubber polymer-coated AC.[22] AC mixed into sediment showed about one order of magnitude weaker sorption than pure AC for HOCs,[13,23] probably attributable to sorptive competition with native HOCs and/or biomolecules or pore clogging.[24] In total, the varied laboratory results demonstrate that the effectiveness of sorbent amendment on lowering contaminant bioavailability increases with decreasing AC particle size, increasing dose of AC, greater mixing, and contact time. Biodynamic modeling with species-specific physiological parameters was able to describe invertebrate tissue concentrations and response to reduced uptake efficiency and pore water concentrations for strongly bound contaminants.[8,23,25] There are many specialty carbons available in the market, but those most suitable for use in sediment remediation will have good sorption properties for the target contaminant (PCBs or Hg for example), will need to have no inherent toxicity, and will need to be low-cost. While some studies[14,22] have compared different types of AC for use in sediment remediation, there is potential for more research in this area.

■ CURRENT STATUS OF TECHNOLOGY DEVELOPMENT: ONGOING PILOT-SCALE DEMONSTRATIONS

Motivated by encouraging bench-scale results, pilot-scale field trials were recently conducted at five sites in the U.S. and Norway as shown in Figure S1 and Table S1 in the Supporting Information. These field experiments are evaluating different methods of applying AC to sediments to reduce the bioavailability of hydrophobic contaminants. The field sites span a range of contaminated aquatic environments: (1) tidal mudflat, (2) freshwater river, (3) marine harbor, (4) deep-water fjord, and (5) tidal creek and marsh. Each site poses varied engineering challenges in the application of AC and monitoring of the effectiveness. The key objectives of the pilot-scale experiments are to study the feasibility of application of AC using large-scale equipment in contaminated field sites, persistence of the AC and its binding potential after application to sediment in the natural environment, effectiveness of the AC in reducing contaminant bioavailability, reductions of sediment porewater contaminant concentrations and sediment-to-water fluxes, and effects of AC addition on the existing benthic community.

A major challenge in pilot evaluations is accounting for transient and/or long-term changes that take place naturally in the open environment. Pilot-studies by design occupy a relatively small footprint in a large contaminated sediment area that typically is overlain by contaminated water mass. Thus, in situ measurements of pore water concentrations at the sediment surface or bioaccumulation assessments using benthic organisms exposed to contaminants in the water phase (e.g., filter-feeding bivalves) can be impacted by the contaminated water above the treatment zone. Finally, over time the small pilot-treatment areas may become covered with newly deposited, contaminated sediment from the surrounding area or upstream locations. Some of the challenges in field assessments can be addressed through appropriate study designs:

(1) Observations of changes in bioaccumulation at treatment sites need to be contrasted to ongoing changes at properly selected background control sites.
(2) Using deposit-feeding organisms for biomonitoring is preferable to using filter feeders for assessing pilot-scale remediation.
(3) In situ assessments should preferably have an ex situ laboratory component to delineate overlying water and depositional impacts.
(4) The number of replicate samplings should be large enough to account for spatial variability at the site.
(5) Multiple lines of evidence for exposure reduction, including physical, chemical, and biological, need to be pursued to obtain confidence in the observations.

■ FINDINGS FROM HUNTERS POINT AND BIG PICTURE

Results from the first pilot study at Hunters Point in San Francisco Bay were recently published.[26,27] The Hunters Point study found that AC can be placed in sediment in a large scale, is physically stable in the environment, and remains effective at binding contaminants in sediments several years after application.[27] The AC applied at Hunters Point did not show a significant impact on benthic community as judged by the diversity of species and their overall abundance. This community-level observation from the field is in contrast to a laboratory study where potential toxic effects of AC on benthic organisms were indicated.[28]

Typical AC dosing at the various test sites was 2−5% by weight of dry sediment (matching the native organic carbon content of sediment) in the top 10−30 cm of sediment. Even under poor mixing conditions, mass transfer of PCBs to a passive sampler in sediment was greatly reduced in the presence of AC.[29] Homogeniety of AC distribution and mixing regime will influence the time required to observe full treatment benefits under field conditions (Figure 3). Small-scale heterogeneity of sorbent distribution at the scale of 1 cm will extend the time required, whereas porewater movement by advection or mechanical dispersion and/or bioturbation will enhance contact between sediment and the added sorbents.

The amount of AC required to remediate a site with 5% in the top 10 cm of bioactive sediment is 35,000 kg/ha which amounts to about \$75,000/ha at a bulk cost of AC of about \$2.2/kg. Cost of AC application will depend on several factors including the need for mixing into sediment, and whether the application and mixing can be accomplished in an exposed sediment surface or needs to be performed underwater. The full cost of AC application is being evaluated through the ongoing pilot studies. By comparison, dredging and disposal cost for the Hudson River cleanup has been projected at \$2.5M/ha [30] and reported actual for phase I at \$15M/ha.[31] Thus, the material cost of AC required for treatment is at least an order of magnitude lower than typical full cost of remediation by dredging and disposal.

The technology is especially attractive at locations where dredging is not feasible or appropriate, such as (i) under piers and around pilings, (ii) in sediment full of debris, (iii) in areas where overdredging is not possible, and (iv) in ecologically sensitive sites such as wetlands. In situ amendments can also be used in combination with other remedies. For example,

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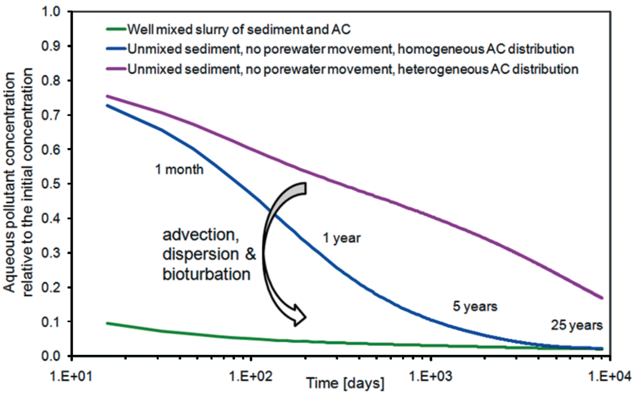

Figure 3. Simulated decrease in the average aqueous PCB-101 concentration for Hunters Point sediment amended with 3.4% by weight activated carbon with a mean particle size of 150 μm. The simulation of the heterogeneous distribution assumes 1 cm spherical volumes of activated carbon free sediment surrounded by activated carbon rich sediment.

sorbent amendments can be applied during and immediately after a dredging process to minimize aqueous contaminant release from resuspended sediments and residuals, or as an amendment to sand caps to enhance retardation capacity.

POTENTIAL USE OF BIOCHARS AND CARBON SEQUESTRATION

Charcoals, especially anthropogenic ones created under high-temperature conditions ("biochar"), are known to persist for thousands of years in soils and sediments, indicating carbon storage opportunities for greenhouse gas abatement.[32,33] AC manufactured from biomass waste products such as pine chips, corn stalk, and poultry litter thus offer an exciting opportunity for efficient resource utilization and carbon sequestration along with sediment remediation.[34] New types of ACs made from renewable resources are being developed and are claimed to have superior metal sorption characteristics.[35] In addition, the U.S. Environmental Protection Agency's new Green Remediation strategy aims to minimize the environmental footprints of a cleanup.[36] Therefore, technologies that can diminish or reverse the carbon footprint while reducing risks will likely be favored in the future. Major unknowns are currently whether a technology can be developed to place (activated) biochars on a sediment bed, and to what extent these materials can be effective in reducing organic and metal contaminant bioavailability in sediments.

POTENTIAL BARRIERS TO USING IN SITU AMENDMENTS AND FUTURE RESEARCH NEEDS

Sorbent amendment does not decrease total sediment concentrations of contaminants. Rather, it decreases contaminants available for biouptake and transport to surface- and groundwater. Sediment risk management is often based on bulk total concentrations and chemical mass with these measures being considered indicative of exposure.[5,37] Although regulatory confidence and comfort are building for the explicit consideration of bioavailability in assessments and remedial decisions, there is still a bias against remedies other than removal. There are also natural perceptions and regulatory precedents to "get it out". This surgical view of sediment remediation is appropriate in many cases but there are numerous situations where removal is not warranted and can be destructive or potentially ineffective for risk reduction. A more balanced evaluation of less invasive remedial measures such as in situ remedies can be achieved by broadening the decision context to include all relevant factors, such as short- and long-term ecological impacts and benefits, residual impacts, and performance. Comparisons of alternatives could involve comparative life cycle assessments.

The pilot studies are starting to provide valuable information to address concerns about long-term effectiveness both in terms of physical stability of the AC and chemical permanence of the remedy. To gain acceptance and advance the technology, it is likely that pilot-scale studies will have to lead to full-scale experimental remedies at a few sites with long-term monitoring to evaluate effectiveness not only near the base of the food chain, but also into evaluating recovery of fish and higher animals that are often the drivers for risk management.

To that end, further research is needed in the following areas:

(1) development of novel amendments that can actively bind contaminants of concern other than HOCs;
(2) improved fundamental understanding of mechanisms of HOC binding to AC, especially in the sediment matrix where fouling can be a concern;
(3) development of efficient, low-impact delivery methods for amendments into sediments;
(4) pilot-scale studies at various hydrodynamic and ecological environments to understand where the technology is best suited;
(5) assessment of ecosystem recovery;
(6) potential for microbial processes to degrade sorbed contaminants
(7) full-scale demonstration to go beyond what can be learned through small-scale pilot studies;
(8) development of modeling tools to interpret field results, understand food web transfer, predict long-term performance, and optimize AC dose and engineering methods of application;
(9) life-cycle analyses including carbon footprints of different sediment remediation technologies.

ASSOCIATED CONTENT

Supporting Information. A summary of ongoing pilot demonstration projects. This information is available free of charge via the Internet at http://pubs.acs.org/.

AUTHOR INFORMATION

Corresponding Author
*E-mail: ughosh@umbc.edu.

BIOGRAPHIES

Upal Ghosh is an associate professor of Civil and Environmental Engineering at the University of Maryland Baltimore County. Richard G. Luthy is Silas H. Palmer Professor in the Department Civil and Environmental Engineering, and Senior Fellow, Woods Institute for the Environment at Stanford University. U.G. and R.G.L. performed initial research on in situ sediment amendments at Stanford University and led the pilot-scale demonstrations at Hunters Point and Grasse River. Gerard Cornelissen is an environmental chemist at the Norwegian Geotechnical Institute, Professor at the University of Life Sciences, Norway, and also attached to Stockholm University, Sweden. He is leading the two AC amendment field studies in

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Norway along with Dr. Espen Eek and NGI-colleagues. Dr. David Werner is a Senior Lecturer in Environmental Engineering in the School of Civil Engineering and Geosciences at Newcastle University and has been closely involved in AC amendment studies especially with respect to contaminant mass transfer and biouptake modeling in AC-amended sediments. Charles A. Menzie is the Director of EcoSciences Practice at Exponent and has expertise in Environmental Risk Assessment. U.G. and C.A.M. codeveloped SediMite through a USEPA SBIR program and are leading the pilot-scale demonstrations in two wetland systems.

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EDITOR'S COMMENTS

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